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27th Annual Report 2018

Convention on Long-range Transboundary Air Pollution

International Cooperative Programme on Integrated Monitoring of Air Pollution Effects on Ecosystems

Sirpa Kleemola and Martin Forsius (eds.)

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wge Working Group on Effects of the Convention on Long-range Transboundary Air Pollution

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ABSTRACT

The Integrated Monitoring Programme (ICP IM) is part of the effect-oriented activities under the 1979 Convention on Long-range Transboundary Air Pollution, which covers the region of the United Nations Economic Commission for Europe (UNECE). The main aim of ICP IM is to provide a framework to observe and understand the complex changes occurring in natural/semi natural ecosystems.

This report summarizes the work carried out by the ICP IM Programme Centre and several collaborating institutes. The emphasis of the report is in the work done during the programme year 2017/2018 including:

- A short summary of previous data assessments
- A status report of the ICP IM activities, content of the IM data base, and geographical coverage of the monitoring network
- A report on long-term changes in the inorganic nitrogen output fluxes in European ICP Integrated Monitoring catchments and an assessment of the role of internal nitrogen parameters
- A progress report on dynamic soil-vegetation modelling
- A literature review: Post disturbance vegetation succession and resilience in forest ecosystems
- National Reports on ICP IM activities are presented as annexes.

Keywords:
Integrated Monitoring, ecosystems, small catchments, air pollution, critical loads, dynamic modelling

TIIVISTELMÄ

Ympäristön yhdennetyn seurannan ohjelma (ICP IM) kuuluu kansainvälisen ilman epäpuhtauksien kaukokulkeutumista koskevan yleissopimuksen ”Convention on Long-range Transboundary Air Pollution” (1979) alaisiin seurantaohjelmiin. Yhdennetyn seurannan ohjelmassa selvitetään kaukokulkeutuvien saasteiden ja muiden ympäristömuutosten vaikutuksia elinympäristöömme. Muutosten seurantaa ja ennusteita muutosten laajuudesta ja nopeudesta tehdään yleensä pienillä metsäisillä valuma-alueilla, mutta verkostoon kuuluu myös muita alueita.

Tämä julkaisu on kooste ohjelmakeskuksen ja yhteistyölaitosten toiminnasta kaudella 2017/2018, joka sisältää:

- Lyhyen yhteenvedon ohjelmassa aiemmin tehdyistä arvioinneista
- Kuvauksen ICP IM ohjelman toiminnasta ja ohjelman seurantaverkosta
- Kuvauksen epäorgaanisen typen huuhtoutumisen pitkän ajan muutoksista ICP IM alueilla ja eri eri valuma-aluetekijöiden vaikutuksista kuormituksen vaihteluun
- Katsauksen dynaamiseen kasvillisuus ja maaperämallintamiseen ICP IM alueilla
- Kirjallisuuskatsauksen liittyen metsäekosysteemien palautumiskykyyn ja kasvillisuusmuutoksiin häiriötilanteiden jälkeen
- Kuvauksia kansallisesta ICP IM toiminnasta eri maissa liitteenä.

Asiasanat:
Yhdennetty ympäristön seuranta, ekosysteemit, pienet valuma-alueet, ilmansaasteet, kriittinen kuormitus, dynaamiset mallit

SAMMANDRAG

Programmet för Integrerad övervakning av miljötillståndet (ICP IM) är en del av monitoringstrategin under UNECE:s luftvårdskonvention (LRTAP). Syftet med ICP IM är att utvärdera komplexa miljöförändringar på avrinningsområden.

Rapporten sammanfattar de utvärderingar som gjorts av ICP IM Programme Centre och de samarbetande instituten under programåret 2017/2018. Rapporten innehåller:

- En sammanfattning av programmets nuvarande omfattning och databasens innehåll
- En syntes av tidigare utvärderingar av data från programmet
- En sammanfattning av modellering av markkemi och markflora
- En rapport om långsiktiga förändringar i flöden av oorganisk kväve från ICP IM områden – inverkan av interna kvävevariabler
- En litteraturöversikt: Tålighet mot förändringar och vegetationssuccession i skogsekosystem
- Beskrivning av nationella ICP IM aktiviteter.

Nyckelord:
Integrerad miljöövervakning, ekosystem, små avrinningsområden, luftföroreningar, kritisk belasting, dynamiska modeller

CONTENTS

ABBREVIATIONS

AMAP	Arctic Monitoring and Assessment Programme
ANC	Acid neutralising capacity
CCE	Coordination Center for Effects
CL	Critical Load
CNTER	Carbon-nitrogen interactions in forest ecosystems
ECE	Economic Commission for Europe
eLTER	The Horizon 2020 project "eLTER" (European Long-Term Ecosystem and socio-ecological Research Infrastructure) A project involving many LTER-Europe partners and sites in collaboration with the European Critical Zone Observatory community
EMEP	Cooperative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe
EU	European Union
EU LIFE	EU's financial instrument supporting environmental and nature conservation projects throughout the EU
Horizon 2020	H2020, EU Research and Innovation programme
ICP	International Cooperative Programme
ICP Forests	International Cooperative Programme on Assessment and Monitoring of Air Pollution Effects on Forests
ICP IM	International Cooperative Programme on Integrated Monitoring of Air Pollution Effects on Ecosystems
ICP Materials	International Cooperative Programme on Effects on Materials
ICP M&M	ICP Modelling and Mapping, International Cooperative Programme on Modelling and Mapping of Critical Loads and Levels and Air Pollution Effects, Risks and Trends
ICP Waters	International Cooperative Programme on Assessment and Monitoring Effects of Air Pollution on Rivers and Lakes
ICP Vegetation	International Cooperative Programme on Effects of Air Pollution on Natural Vegetation and Crops
ILTER	International Long Term Ecological Research Network
IM	Integrated Monitoring
JEG	JEG DM, Joint Expert Group on Dynamic Modelling
LRTAP Convention	Convention on Long-range Transboundary Air Pollution
LTER-Europe	European Long-Term Ecosystem Research Network
LTER-Network	Long Term Ecological Research Network
NFP	National Focal Point
TF	Task Force
Task Force on Health	The Joint Task Force on the Health Aspects of Air Pollution
UNECE	United Nations Economic Commission for Europe
WGE	Working Group on Effects

Summary

Background and objectives of ICP IM

Integrated monitoring of ecosystems means physical, chemical and biological measurements over time of different ecosystem compartments simultaneously at the same location. In practice, monitoring is divided into a number of compartmental sub-programmes which are linked by the use of the same parameters (cross-media flux approach) and/or same or close stations (cause-effect approach).

The International Cooperative Programme on Integrated Monitoring of Air Pollution Effects on Ecosystems (ICP IM, www.syke.fi/nature/icpim) is part of the Effects Monitoring Strategy under the Convention on Long-range Transboundary Air Pollution (LRTAP Convention). The main objectives of the ICP IM are:

- To monitor the biological, chemical and physical state of ecosystems (catchments/plots) over time in order to provide an explanation of changes in terms of causative environmental factors, including natural changes, air pollution and climate change, with the aim to provide a scientific basis for emission control.
- To develop and validate models for the simulation of ecosystem responses and use them (a) to estimate responses to actual or predicted changes in pollution stress, and (b) in concert with survey data to make regional assessments.
- To carry out biomonitoring to detect natural changes, in particular to assess effects of air pollutants and climate change.

The full implementation of the ICP IM will allow ecological effects of heavy metals, persistent organic substances and tropospheric ozone to be determined. A primary concern is the provision of scientific and statistically reliable data that can be used in modelling and decision making.

The ICP IM sites (mostly forested catchments) are located in undisturbed areas, such as natural parks or comparable areas. The ICP IM network presently covers fifty sites from sixteen countries. The international Programme Centre is located at the Finnish Environment Institute in Helsinki. The present status of the monitoring activities is described in detail in Chapter 1 of this report.

A manual detailing the protocols for monitoring each of the necessary physical, chemical and biological parameters is applied throughout the programme (Manual for Integrated Monitoring 1998, and updated web version).

Assessment activities within the ICP IM

Assessment of data collected in the ICP IM framework is carried out at both national and international levels. Key tasks regarding international ICP IM data have been:

- Input-output and proton budgets
- Trend analysis of bulk and throughfall deposition and runoff water chemistry
- Assessment of responses using biological data
- Dynamic modelling and assessment of the effects of different emission / deposition scenarios, including confounding effects of climate change processes
- Assessment of concentrations, pools and fluxes of heavy metals
- Calculation of critical loads for sulphur and nitrogen compounds, and assessment of critical load exceedance, as well as links between critical load exceedance and empirical impact indicators.

Conclusions from international studies using ICP IM data

Input-output and proton budgets, C/N interactions

Ion mass budgets have proved to be useful for evaluating the importance of various biogeochemical processes that regulate the buffering properties in ecosystems. Long-term monitoring of mass balances and ion ratios in catchments/plots can also serve as an early warning system to identify the ecological effects of different anthropogenically derived pollutants, and to verify the effects of emission reductions.

The most recent results from ICP IM studies are available from the study of Vuorenmaa et al. (2017). Site-specific annual input-output budgets were calculated for sulphate (SO_4) and total inorganic nitrogen (TIN = NO_3-N + NH_4-N) for 17 European ICP IM sites in 1990–2012. Temporal trends for input (deposition) and output (runoff water) fluxes and net retention/net release of SO_4 and TIN were also analysed. Large spatial variability in the input and output fluxes of SO_4 and TIN reflects important gradients of air pollution effects in Europe, with the highest deposition and runoff water fluxes in southern Scandinavia, Central and Eastern Europe and the lowest fluxes at more remote sites in northern European regions. A significant decrease in the total (wet + dry) non-marine SO_4 deposition and bulk deposition of TIN was found at 90% and 65% of the sites, respectively. Output fluxes of non-marine SO_4 in runoff decreased significantly at 65% of the sites, indicating positive effects of international emission abatement actions in Europe during the last 25 years. Catchments retained SO_4 in the early and mid-1990s, but this shifted towards a net release in the late 1990s, which may be due to the mobilization of legacy S pools accumulated during times of high atmospheric SO_4 deposition. Despite decreased deposition, TIN output fluxes and retention rates showed a mixed response with both decreasing (9 sites) and increasing (8 sites) trend slopes, but trends were rarely significant. In general, TIN was strongly retained in the catchments not affected by natural disturbances. The long-term annual variation in net releases for SO_4 was explained by variations in runoff and SO_4 concentrations in deposition, while a variation in TIN concentrations in runoff was mostly associated with a variation of the TIN retention rate in catchments. Net losses of SO_4 may lead to a slower recovery of surface waters than those predicted by the decrease in SO_4 deposition. Continued enrichment of N in catchment soils poses a threat to terrestrial biodiversity and may ultimately lead to higher TIN runoff through N saturation or climate change. Continued monitoring and further evaluations of mass balance budgets are thus needed.

Earlier results from ICP IM studies are summarized below.

The first results of input-output and proton budget calculations were presented in the 4th Annual Synoptic Report (ICP IM Programme Centre 1995) and the updated results regarding the effects of N deposition were presented in Forsius et al. (1996). Data from selected ICP IM sites were also included in European studies for evaluating soil organic horizon C/N-ratio as an indicator of nitrate leaching (Dise et al. 1998, MacDonald et al. 2002). Results regarding the calculation of fluxes and trends of S and N compounds were presented in a scientific paper prepared for the Acid Rain Conference, Japan, December 2000 (Forsius et al. 2001). A scientific paper regarding calculations of proton budgets was published in 2005 (Forsius et al. 2005).

The budget calculations showed that there was a large difference between the sites regarding the relative importance of the various processes involved in the transfer of acidity. These differences reflected both the gradients in deposition inputs and the differences in site characteristics. The proton budget calculations showed a clear relationship between the net acidifying effect of nitrogen processes and the amount of N deposition. When the deposition increases also N processes become increasingly important as net sources of acidity.

A critical deposition threshold of about 8–10 kg N ha^{-1} yr^{-1}, indicated by several previous assessments, was confirmed by the input-output calculations with the ICP IM data (Forsius et al. 2001). The output flux of nitrogen was strongly correlated with key ecosystem variables like N deposition, N concentration in organic matter and current year needles, and N flux in litterfall (Forsius et al. 1996). Soil organic horizon C/N-ratio seems to give a reasonable estimate of the annual export flux of N for European forested sites receiving throughfall deposition of N up to about 30 kg N ha^{-1} yr^{-1}. When stratifying data based on C/N ratios less than or equal to 25 and greater than 25, highly significant relationships were observed between N input and nitrate leached (Dise et al. 1998, MacDonald et al. 2002, Gundersen et al. 2006). Such statistical relationships from intensively studied sites can be efficiently used in conjugation with regional monitoring data (e.g. ICP Forests and ICP Waters data) in order to link process level data with regional-scale questions.

An assessment on changes in the retention and release of S and N compounds at the ICP IM sites was prepared for the 21st Annual Report (Vuorenmaa et al. 2012). Updated and revised data were included in the continuation of the work in the 22nd and 23rd Annual Reports (Vuorenmaa et al. 2013, 2014). The relationship between N deposition and organic N loss and the role of organic nitrogen in the total nitrogen output fluxes were derived in Vuorenmaa et al. (2013).

Sulphur budgets calculations indicated a net release of S from many ICP IM sites, indicating that the soils are releasing previously accumulated S. Similar results have been obtained in other recent European plot and catchment studies.

The reduction in deposition of S and N compounds at the ICP IM sites, caused by the "Protocol to Abate Acidification, Eutrophication and Ground-level Ozone" of the LRTAP Convention ("Gothenburg protocol"), was estimated for the year 2010 using transfer matrices and official emissions. Implementation of the protocol will further decrease the deposition of S and N at the ICP IM sites in western and north western parts of Europe, but in more eastern parts the decrease will be smaller (Forsius et al. 2001).

Results from the ICP IM sites were also summarised in an assessment report prepared by the Working Group on Effects of the LRTAP Convention (WGE) (Sliggers & Kakebeeke 2004, Working Group on Effects 2004).

ICP IM contributed to an assessment report on reactive nitrogen (N_r) of the WGE. This report was prepared for submission to the TF on Reactive Nitrogen and other bodies of the LRTAP Convention to show what relevant information has been collected by the ICP programmes under the aegis of the WGE to allow a better understanding of N_r effects in the ECE region. The report contributed relevant information for the revision of the Gothenburg Protocol. A revised Gothenburg Protocol was successfully finalised in 2012.

It should also be recognized that there are important links between N deposition and the sequestration of C in the ecosystems (and thus direct links to climate change processes). These questions were studied in the CNTER-project in which data from both the ICP IM and EU/Intensive Monitoring sites were used (Gundersen et al. 2006). A summary report of the CNTER-results on C/N -interactions and nitrogen effects in European forest ecosystems was prepared for the WGE meeting 2007 (ECE/EB.AIR/WG.1/2007/10).

Trend analysis

Empirical evidence on the development of environmental effects is of central importance for the assessment of success of international emission reduction policy. In order to assess the impacts of air pollution and climate change in the environment, a long-term integrated monitoring approach in remote unmanaged areas including physical, chemical and biological variables is needed. Vuorenmaa et al. (2018) evaluated long-term trends (1990–2015) for deposition and runoff water chemistry and fluxes, and climatic variables at 25 ICP IM sites in Europe that commonly belong also to the LTER-Europe/ILTER networks. The trend assessment was published in a special issue in Science of the Total Environment with a working title: "Detecting and explaining natural and anthropogenic changes by making use of large extent, long-term ecological research facilities of the international long-term ecosystem research (ILTER) network". The recent results from trend assessment at IM sites confirm that emission abatement actions are having their intended effects on precipitation and runoff water chemistry in the course of successful emission reductions in different regions in Europe. Concentrations and deposition fluxes of xSO_4, and consequently acidity in precipitation, have substantially decreased in IM areas. Inorganic N (TIN) deposition has decreased in most of the IM areas, but to a lesser extent than that of xSO_4. Substantially decreased xSO_4 deposition has resulted in decreased concentrations and output fluxes of xSO_4 in runoff, and decreasing trends of TIN concentrations in runoff – particularly for NO_3 – are more prominent than increasing trends. In addition, decreasing trends appeared to strengthen over the course of emission reductions during the last 25 years. TIN concentrations in runoff were mainly decreasing, while trends in output fluxes were more variable, but trend slopes were decreasing rather than increasing. However, decreasing trends for S and N emissions and deposition and deposition reduction responses in runoff water chemistry tended to be more gradual since the early 2000s. Air temperature increased significantly at 61% of the sites, while trends for precipitation and runoff were rarely significant. The site-specific variation of xSO_4 concentrations in runoff was most strongly explained by deposition. Climatic variables and deposition explained the variation of TIN concentrations in runoff at single sites poorly, and as yet there are no clear signs of a consistent deposition-driven or climate-driven increase in TIN exports in the catchments.

Vuorenmaa et al. (2018) reported that the IM sites are located in areas with very different N deposition gradients, and it is obvious that not all potential drivers were included in the empirical model in the study, and further analysis with specific landscape and soil data is needed to elucidate the variation in inorganic N concentrations in runoff at IM sites. Thus, the next phase of the work on trend assessment will be an assessment of the role of internal nitrogen parameters (see Chapter 3 in this report, Vuorenmaa et al.).

Earlier work is summarized below.

First results from a trend analysis of monthly ICP IM data on bulk and throughfall deposition as well as runoff water chemistry were presented in Vuorenmaa (1997). ICP IM data on water chemistry were also used for a trend analysis carried out by the ICP Waters and results were presented in the Nine Year Report of that programme (Lükewille et al. 1997).

Calculations on the trends of N and S compounds, base cations and hydrogen ions were made for 22 ICP IM sites with available data across Europe (Forsius et al. 2001). The site-specific trends were calculated for deposition and runoff water fluxes using monthly data and non-parametric methods. Statistically significant downward trends of SO_4, NO_3 and NH_4 bulk deposition (fluxes or concentrations) were observed at

50% of the ICP IM sites. Sites with higher N deposition and lower C/N-ratios clearly showed higher N output fluxes, and the results were consistent with previous observations from European forested ecosystems. Decreasing SO_4 and base cation trends in runoff waters were commonly observed at the ICP IM sites. At some sites in the Nordic countries decreasing NO_3 and H^+ trends (increasing pH) were also observed. The results partly confirmed the effective implementation of emission reduction policy in Europe. However, clear responses were not observed at all sites, showing that recovery at many sensitive sites can be slow and that the response at individual sites may vary greatly.

Data from ICP IM sites were also used in a study of the long-term changes and recovery at nine calibrated catchments in Norway, Sweden and Finland (Moldan et al. 2001, RECOVER: 2010 project). Runoff responses to the decreasing deposition trends were rapid and clear at the nine catchments. Trends at all catchments showed the same general picture as from small lakes in Scandinavia.

It was agreed at the ICP IM Task Force meeting in 2004 that a new trend analysis should be carried out. The preliminary results were presented in Kleemola (2005) and the updated results in the 15th Annual Report (Kleemola & Forsius 2006). Statistically significant decreases in SO_4 concentrations were observed at a majority of sites in both deposition and runoff/soil water quality. Increases in ANC (acid neutralising capacity) were also commonly observed. For NO_3 the situation was more complex, with fewer decreasing trends in deposition and even some increasing trends in runoff/soil water.

Results from several ICPs and EMEP were used in an assessment report on acidifying pollutants, arctic haze and acidification in the arctic region prepared for the Arctic Monitoring and Assessment Programme (AMAP, Forsius & Nyman 2006, www.amap.no). Sulphate concentrations in air showed generally decreasing trends since the 1990s. In contrast, levels of nitrate aerosol were increasing during the arctic haze season at two stations in the Canadian arctic and Alaska, indicating a decoupling between the trends in sulphur and nitrogen. Chemical monitoring data showed that lakes in the Euro-Arctic Barents region are showing regional scale recovery. Direct effects of sulphur dioxide emissions on trees, dwarf shrubs and epiphytic lichens were observed close to large smelter point sources.

The recent trend assessment using monthly ICP IM data (Vuorenmaa et al. 2018) was preceded by corresponding trend evaluations for the periods 1993–2006 and 1990–2013 (Vuorenmaa et al. 2009, 2016, respectively). Moreover, trends for annual input and output fluxes of SO_4 and TIN were evaluated for the period 1990–2012 (Vuorenmaa et al. 2017). These results clearly showed the regional-scale decreasing trends of SO_4 in deposition and runoff/soil water, and suggested that IM catchments have increasingly responded to the decreases in S emissions and depositions of SO_4 since the early 1990s. Decreased nitrogen emissions also resulted in decrease of inorganic N deposition, but to a lesser extent than that of SO_4, and trends in TIN fluxes in runoff were highly variable due to complex processes in terrestrial catchment that are not yet fully understood. Besides, the net release of SO_4 in forested catchments fueled by the mobilization of legacy S pools, accumulated during times of high atmospheric sulphur deposition, may delay the recovery from acidification. The more efficient retention of inorganic N than SO_4 results in generally higher leaching fluxes of SO_4 than those of inorganic N in European forested ecosystems. SO_4 thus remains the dominant source of actual soil acidification despite the generally lower input of SO_4 than inorganic N. Critical load calculations for Europe also indicated exceedances of the N critical loads over large areas. Long-term trends for deposition and runoff variables were for the first time evaluated together with climatic variables (precipitation, runoff water volume and air temperature) at IM sites by Vuorenmaa et al. (2016). Many study sites exhibited long-term seasonal trends with a significant increase in air temperature, precipitation and runoff particularly in spring and autumn, but annual trends were rarely significant. It was concluded that the sulphur and nitrogen problem thus clearly requires continued attention as a European air pollution issue, and further long-term monitoring and trend assessments of different ecosystem compartments and climatic variables are needed to evaluate the effects, not only of emission reduction policies, but also of changing climate.

An assessment on changes in the retention and release of S and N compounds at the ICP IM sites was prepared for the 21st Annual Report (Vuorenmaa et al. 2012). Updated and revised data were included in the continuation of the work in the 22nd and 23rd Annual Reports. The role of organic nitrogen in mass balance budget was derived and trends of S and N in fluxes were analysed (Vuorenmaa et al. 2013, 2014).

Detected responses in biological data

The effect of pollutant deposition on natural vegetation, including both trees and understorey vegetation, is one of the central concerns in the impact assessment and prediction. The most recent ICP IM study on dose-response relationships was published by Dirnböck et al. (2014). This study utilized a new ICP IM database for biological data and focussed on effects on forest floor vegetation from elevated nitrogen deposition.

In many European countries airborne nitrogen coming from agriculture and fossil fuel burning exceeds critical thresholds and threatens the functioning of ecosystems. One effect is that high levels of nitrogen stimulate the growth of only a few plants which outcompete other, often rare species. As a consequence biodiversity declines. Though this is known to happen in natural and semi-natural grasslands, it has never been shown in forest ecosystems where management is a strong, mostly overriding determinant of biodiversity. Dirnböck et al. (2014) utilized long-term monitoring data from 28 Integrated Monitoring sites to analyse temporal trends in plant species cover and diversity. At sites where nitrogen deposition exceeded the critical load, the cover of forest plant species preferring nutrient-poor soils (oligotrophic species) significantly decreased whereas plant species preferring nutrient-rich soils (eutrophic species) showed – though weak – an opposite trend. These results show that airborne nitrogen has changed the structure and composition of forest floor vegetation in Europe. Plant species diversity did not decrease significantly within the observed period but the majority of newly established species was found to be eutrophic. Hence it was hypothesized that without reducing nitrogen deposition below the critical load forest biodiversity will decline in the future.

Previous work on biological data is summarized below.

The first assessment of vegetation monitoring data at ICP IM sites with regards to N and S deposition was carried out by Liu (1996). Vegetation monitoring was found useful in reflecting the effects of atmospheric deposition and soil water chemistry, especially regarding sulphur and nitrogen. The results suggested that plants respond to N deposition more directly than to S deposition with respect to vegetation indices.

De Zwart (1998) carried out an exploratory multivariate statistical gradient analysis of possible causes underlying the aspect of forest damage at ICP IM sites. These results suggested that coniferous defoliation, discolouration and lifespan of needles in the diverse phenomena of forest damage are for respectively 18%, 42% and 55% explained by the combined action of ozone and acidifying sulphur and nitrogen compounds in air.

As a separate exercise, the epiphytic lichen flora of 25 European ICP IM monitoring sites, all situated in areas remote from local air pollution sources, was statistically related to measured levels of SO_2 in air, NH_4^+, NO_3^- and SO_4^{2-} in precipitation, annual bulk precipitation, and annual average temperature (van Herk et al. 2003, de Zwart et al. 2003). It was concluded that long distance transport of nitrogen air pollution is important in determining the occurrence of acidophytic lichen species, and constitutes a threat to natural populations that is strongly underestimated so far.

In 2010, the Task Force meeting decided upon a new reporting format for biological data. The new format was based on primary raw data, and not aggregated mean values as before. All countries were encouraged to re-report old data in the new format. This was successful and as a result, the full potential of the biological data from the ICP Integrated Monitoring network could be utilised to raise and answer research question that the old database could not.

Dynamic modelling and assessment of the effects of emission/deposition scenarios

In a policy-oriented framework, dynamic models are needed to explore the temporal aspect of ecosystem protection and recovery. The critical load concept, used for defining the environmental protection levels, does not reveal the time scales of recovery. Priority in the ICP IM work is given to site-specific modelling. The role of ICP IM is to provide detailed and consistent physical and chemical data and long time-series of observations for key sites against which model performance can be assessed and key uncertainties identified (see Jenkins et al. 2003). ICP IM participates also in the work of the Joint Expert Group on Dynamic Modelling (JEG) of the WGE.

Dynamic vegetation modelling at ICP IM sites has been initiated with contributions from ICP M&M, ICP Forests, and the LTER Europe network. The VSD+ model was applied to simulate soil chemistry at 26 sites in ten countries throughout Europe (Holmberg & Dirnböck 2015, 2016, Holmberg et al. 2018). Simulated future soil conditions improved under projected decrease in deposition and current climate conditions: higher pH, BS and C:N at 21, 16 and 12 of the sites, respectively. Work is ongoing regarding modelling forest vegetation response (see Chapter 2 in this report, Dirnböck et al.).

Dynamic models have also previously been developed and used for the emission/deposition and climate change scenario assessment at several selected ICP IM sites (e.g. Forsius et al. 1997, 1998a, 1998b, Posch et al. 1997, Jenkins et al. 2003, Futter et al. 2008, 2009). These models are flexible and can be adjusted for the assessment of alternative scenarios of policy importance. The modelling studies have shown that the recovery of soil and water quality of the ecosystems is determined by both the amount and the time of implementation of emission reductions. According to the models, the timing of emission reductions determines the state of recovery over a short time scale (up to 30 years). The quicker the target level of reductions is achieved, the more rapidly the surface water and soil status recover. For the long-term response (> 30 years), the magnitude of emission reductions is more important than the timing of the reduction. The model simulations also indicate that N emission controls are very important to enable the maximum recovery in response to S emission reductions. Increased nitrogen leaching has the potential to not only offset the recovery predicted in response to S emission reduction, but further to promote substantial deterioration in pH status of freshwaters and other N pollution problems in some areas of Europe. Work has also been conducted to predict potential climate change impacts on air pollution related processes at the sites. The large EU-project Euro-limpacs (2004–2009) studied the global change impacts on freshwater ecosystems. The institutes involved in the project used data collected at ICP IM and ICP Waters sites as key datasets for the modelling, time-series and experimental work of the project. A modelling assessment on the global change impacts on acidification recovery was carried out in the project (Wright et al. 2006). The results showed that climate/global change induced changes may clearly have a large impact on future acidification recovery patterns, and need to be addressed if reliable future predictions are wanted (decadal time scale). However, the relative significance of the different scenarios was to a large extent determined by site-specific characteristics. For example, changes in sea-salt deposition were only important at coastal sites and changes in decomposition of organic matter at sites which are already nitrogen saturated.

In response to environmental concerns, the use of biomass energy has become an important mitigation strategy against climate change. A summary report on links between climate change and air pollution effects, based on results of the Euro-limpacs project, was prepared for the WGE meeting 2008 (ECE/EB.AIR/WG.1/2008/10). It was concluded that the increased use of forest harvest residues for biofuel production is predicted to have a significant negative influence on the base cation budgets causing re-acidification at the study catchments. Sustainable forestry management policies would need to consider the combined impact of air pollution and harvesting practices.

Pools and fluxes of heavy metals

The work to assess concentrations, stores and fluxes of heavy metals at ICP IM sites is led by Sweden. In 26th Annual Report data on Pb, Cd, Hg, Cu and Zn from countries in the ICP IM were presented (Åkerblom & Lundin 2017). These data will be used for establishment of background heavy metal concentrations in forested compartments and risk assessments of heavy metals. In many national studies on ICP IM sites, detailed site-specific budget calculations of heavy metals (including Hg) have improved the scientific understanding of ecosystem processes, retention times and critical thresholds. ICP IM sites are also used for dynamic model development of these compounds.

For the future evaluation of emission reductions of heavy metals to the atmosphere we will analyze site-specific long-term trends for fluxes of heavy metals (primarily for Cd, Pb, and Hg and depending on availability of data, also Cu and Zn) in deposition (input) and runoff (output), using available long-term monthly data collected across ICP IM sites in Europe. This will be done to see if fluxes of heavy metals in deposition and runoff respond to changes in emission reductions in Europe. Reduction in heavy metal emissions is hypothesized to be reflected in decreasing heavy metal concentrations (Åkerblom & Lundin 2015), taking into account climatic variation over time and between regions also in decreasing heavy metal fluxes. Temporal trend analysis in heavy metal fluxes will provide a detailed understanding of responses in heavy metal mass balances to emission reductions and give indication on possible change in retention of heavy metals in catchments over time. This overview will also provide an estimate on the significance in heavy metal mass balances over time and identify uncertainties in the mass balances and needs for improvements. The aim is to present results in an international scientific journal.

Increases or no change in Hg concentrations in the upper-most forest soil mor-layer does not correspond to the general decrease in emissions of Hg to the atmosphere and atmospheric concentrations (Åkerblom & Lundin 2015). Apparently there is an insufficient understanding of the governing processes and a need for more detailed data on the Hg cycling in forest catchments. One process that is not accounted for in ICP IM programme is the land-atmosphere exchange of Hg. The phenomena of land-atmosphere exchange has been known for long time but it has been quantified only recently due to the development of micrometerological systems for continuous measurements (Osterwalder et al. 2016). In the case of mass balance calculations for Hg new evidence has shown that land-atmosphere exchange during a 2-year study over a peatland can be more than double the flux in stream runoff (Osterwalder et al. 2017). Based on natural Hg stable isotope studies in podzols and histosols, significant

Hg re-emission from organic soil horizons occurred (Jiskra et al. 2015). These novel observations and knowledge about processes that govern land-atmosphere exchange of Hg calls for methods and approaches to account for this important flux in the catchment cycle of Hg within ICP IM.

Previous work on heavy metals is summarized below.

Preliminary results on concentrations, fluxes and catchment retention were reported to the Working Group on Effects in 2001 (document EB.AIR/WG.1/2001/10). The main findings on heavy metals budgets and critical loads at ICP IM sites were presented by Bringmark (2011). Input/output budgets and catchment retention for Cd, Pb and Hg in the years 1997–2011 were determined for 14 ICP IM catchments across Europe (Bringmark et al. 2013). Litterfall plus throughfall was taken as a measure of the total deposition of Pb and Hg (wet + dry) on the basis of evidence suggesting that, for these metals, internal circulation is negligible. The same is not true for Cd. Excluding a few sites with high discharge, between 74 and 94 % of the input, Pb was retained within the catchments; significant Cd retention was also observed. High losses of Pb (>1.4 $mg\,m^{-2}\,yr^{-1}$) and Cd (>0.15 $mg\,m^{-2}\,yr^{-1}$) were observed in two mountainous Central European sites with high water discharge. All other sites had outputs below or equal to 0.36 and 0.06 $mg\,m^{-2}\,yr^{-1}$, respectively, for the two metals. Almost complete retention of Hg, 86–99 % of input, was reported in the Swedish sites. These high levels of metal retention were maintained even in the face of recent dramatic reductions in pollutant loads. In the Progress report on heavy metal trends at ICP IM sites (Åkerblom & Lundin 2015) temporal trends were seen in forest floor with decreasing concentrations for Cd and Pb while Hg did not change. An increase in heavy metal concentrations was also seen in deeper mineral soil horizon indicating a translocation of heavy metals from upper to deeper soil horizons.

Calculation of critical loads and their exceedance, relationships to effect indicators

Empirical impact indicators of acidification and eutrophication were determined from stream water chemistry and runoff observations at ICP IM catchments (Holmberg et al. 2013). The indicators were compared with exceedances of critical loads of acidification and eutrophication obtained with deposition estimates for the year 2000. Empirical impact indicators agreed well with the calculated exceedances. Annual mean fluxes and concentrations of acid neutralizing capacity (ANC) were negatively correlated with the exceedance of critical loads of acidification. Observed leaching of nitrogen was positively correlated with the exceedances of critical loads (Holmberg et al. 2013). This study was revisited with new data on N concentrations and fluxes (Holmberg et al. 2017). For most sites, there was an improvement visible as a shift towards less exceedance and lower concentrations of total inorganic nitrogen (TIN) in runoff. At the majority of the sites both the input and the output flux of TIN decreased between the two observation periods 2000–2002 and 2013–2015. Data from the ICP IM provide evidence of a connection between modelled critical loads and empirical monitoring results for acidification parameters and nutrient nitrogen.

Planned activities

- Maintenance and development of a central ICP IM database at the Programme Centre.
- Continued assessment of the long-term effects of air pollutants to support the implementation of emission reduction protocols, including:
  - Assessment of trends.
  - Calculation of ecosystem budgets, empirical deposition thresholds and site-specific critical loads.
  - Dynamic modelling and scenario assessment.
  - Comparison of calculated critical load exceedances with observed ecosystem effects.
- Calculation of pools and fluxes of heavy metals at selected sites.
- Assessment of cause-effect relationships for biological data, particularly vegetation.
- Coordination of work and cooperation with other ICPs, particularly regarding dynamic modelling (all ICPs), cause-effect relationships in terrestrial systems (ICP Forests, ICP Vegetation), and surface waters (ICP Waters).
- Participation in the development of the European LTER-network (Long Term Ecological Research network, www.lter-europe.net), and the related EU/H2020-infrastructure project eLTER.
- Cooperation with other external organisations and programmes, particularly the International Long Term Ecological Research network (ILTER, www.ilter.network).
- Participation in projects with a global change perspective.

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1 ICP IM activities, monitoring sites and available data

1.1 Review of the ICP IM activities in 2017–2018

Meetings

- The former ICP IM Chair Lars Lundin represented ICP IM in the 33rd Task Force Meeting of ICP Forests in Bucharest, Romania, 15–18 May 2017.
- ICP IM Programme Manager Martin Forsius and Chair Ulf Grandin participated in the eLTER H2020 project annual progress meeting in Vienna, Austria, 7–9 June 2017.
- Ulf Grandin, Martin Forsius and Lars Lundin represented ICP IM in the Third Joint session of the Working Group on Effects and the Steering Body to EMEP in Geneva, Switzerland, 11–15 September 2017.
- Martin Forsius participated in the joint 25th annual ILTER meeting & LTER France meeting in Nantes, France, 2–6 October 2017.
- Martin Forsius, Ulf Grandin, Thomas Dirnböck and Maria Holmberg took part in the eLTER Conference organized back-to-back with the annual LTER Europe business meeting in Malaga, Spain from 28 November to 1 December 2017.
- Maria Holmberg and Thomas Dirnböck gave a joint presentation "Observing and modelling long-term effects of reactive Nitrogen in ecosystems" on 28 November 2017 at the Science Day of the eLTER Conference in Malaga, Spain.
- New Co-Chairs Ulf Grandin and Salar Valinia, and Martin Forsius represented ICP IM in the EMEP Steering Body and Working Group on Effects Bureaux meeting in Madrid, Spain, 19–21 February 2018.
- Martin Forsius participated in the Core team of eLTER (H2020), 1–2 March 2018, and the LTSER workshop, 4–8 March 2018, organized in Ktura, Israel.
- Ulf Grandin, Martin Forsius and Salar Valinia took part in the sixth "Saltsjöbaden" workshop in Gothenburg, Sweden, 19-21 March 2018.
- Thomas Dirnböck (Austria) represented ICP IM in the 33rd Task Force of the ICP Modelling & Mapping meeting and Joint Expert Group on Dynamic Modelling in Bern, Switzerland, 18–20 April 2018.
- The twenty-sixth meeting of the Programme Task Force on ICP Integrated Monitoring was organized as a joint 2018 Task Force Meeting of ICP Waters and ICP Integrated Monitoring in Warsaw, Poland from May 7 to May 9, 2018.

Projects, data issues

After December 1st 2017 the National Focal Points (NFPs) reported their 2016 results to the ICP IM Programme Centre. The Programme Centre carried out standard check-up of the results and incorporated them into the IM database.

Scientific work in priority topics

- The Programme Centre prepared the ICP IM contribution to the Joint Report 2017 of the ICPs, TF health and Joint Expert Group on Dynamic Modelling for the WGE (ECE/EB.AIR/GE.1/2017/3- ECE/EB.AIR/WG.1/2017/3).
- Scientific paper "Long-term changes (1990–2015) in the atmospheric deposition and runoff water chemistry of sulphate, inorganic nitrogen and acidity for forested catchments in Europe in relation to changes in emissions and hydrometeorological conditions" (J. Vuorenmaa et al. 2018) was finalized and was published in Science of The Total Environment.
- Scientific paper "Modelling study of soil C, N and pH response to air pollution and climate change using European LTER site observations" (M. Holmberg et al. 2018) was finalized and published in the ILTER special issue in Science of The Total Environment: "Detecting and explaining natural and anthropogenic changes by making use of large extent, long-term ecological research facilities of the international long-term ecosystem research (ILTER) network".
- ICP IM has contributed to the Joint Report on mercury in the aquatic environment (Joint report together with ICP Waters) (Braaten et al. 2017).
- ICP IM participates in a joint coordinated exercise on dynamic modelling together with other ICPs (Joint Expert Group on Dynamic Modelling, JEG DM). Priority in the ICP IM work is given to site-specific modelling activities and development/testing of new methodologies for assessing the connections between air pollution and climate change.

1.2 Activities and tasks planned for 2019–2020

Activities/tasks related to the programme's present objectives, carried out in close collaboration with other ICPs/ Task Forces

According to the ICP IM work plan, ICP IM will produce the following reports:

2019: Report on dynamic modelling on the impacts of deposition and climate change scenarios on ground vegetation

2019: Scientific paper on the relationship between critical load exceedances and empirical ecosystem impact indicators

2019: Scientific paper on heavy metal trends in concentrations and fluxes across ICP IM sites in Europe, in cooperation with ICP Waters

2019: Scientific paper on the impacts of catchment characteristics, climate and hydrology on N processes

Other activities

- Maintenance and development of central ICP IM database at the Programme Centre
- Arrangement of the 27th Task Force meeting (2019)
- Preparation of the 28th ICP IM Annual Report (2019)
- Preparation of the ICP IM contribution to assessment reports of the WGE
- Participation in meetings of the WGE, other ICPs and the JEG DM

Activities/tasks aimed at further development of the programme

- Participation in the development of the European LTER-network (Long Term Ecological Research network, www.lter-europe.net), and the EU/H2020 eLTER-project.
- Participation in the activities of other external organisations, particularly the International Long Term Ecological Research Network (ILTER, www.ilter.network)

1.3 Published reports and articles 2017–2018

Evaluations of international ICP IM data and related publications

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Kleemola, S. & Forsius, M. (Eds.) 2017. 26th Annual Report 2017. Convention on Long-range Transboundary Air Pollution, ICP Integrated Monitoring. Reports of the Finnish Environment Institute 24/2017, Finnish Environment Institute, Helsinki. 70 p. http://hdl.handle.net/10138/212199

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Evaluations of national ICP IM data and publications of ICP IM representatives

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1.4 Monitoring sites and data

The following countries have continued data submission to the ICP IM data base during the period 2013–2017: Austria, Belarus, the Czech Republic, Estonia, Finland, Germany, Ireland, Italy, Lithuania, Norway, Poland, the Russian Federation, Spain, Sweden, Switzerland and Ukraine. Poland rejoined the network and included several sites in 2017.

The number of sites with on-going data submission for at least part of the data years 2012–2016 is 50 from sixteen countries. Sites from Canada, Latvia and United Kingdom only contain older data.

An overview of the data reported internationally to the ICP IM database is given in Table 1.1. Additional earlier reported data are available from sites outside those presented in Table 1.1. and Fig. 1.1. Locations of the ICP IM monitoring sites are shown in Fig. 1.1.

Table 1.1. Internationally reported data from ICP IM sites (- subprogramme not possible to carry out, * or forest health parameters in former Forest stands/Trees).

AREA		SUBPROGRAMME*																							
		AM	AC	PC	MC	TF	SF	SC	SW	GW	RW	LC	FC	LF	RB	LB	FD	VG	BI	VS	EP	AL	MB	BB	BV
		meteorology	air chemistry	precipitation chemistry	moss chemistry	throughfall	stemflow	soil chemistry	soil water chemistry	groundwater chemistry	runoff water chemistry	lake water chemistry	foliage chemistry	litterfall	hydrobiology of streams	hydrobiology of lakes	forest damage	vegetation	bioelements	vegetation structure	trunk epiphytes	aerial green algae	microbial decomposition	bird inventory	vegetation inventory
AT01	ZÖBELBODEN	95-16	95-16	93-16		93-16	99-04		93-16		95-16	-	92-11	93-16				93			93-98				
BY02	BEREZINA BR	89-15	89-15	89-15				95-98			95-15														
CH02	LAGO NERO	15	15-16	15-16							15-16	15-16													
CZ01	ANENSKE POVODI	89-16	89-16	89-16	89	89-16			07-16	08-16	89-16	-			07	-									
CZ02	LYSINA	67-16	93-96	90-16		91-16		93	90-16	89-16	89-16	91-16	94	08	07	11		15	94			14-15		10	
DE01	FORELLENBACH	90-16	90-16	90-16	90	90-16	90-05	90-11	90-16	88-16	90-16	-	90-16	90-16		-	90-14	90-08		00	92-95		94-16	91-02	90-95
DE02	NEUGLOBSOW	67-16	98-16	98-16		98-16	04-16	04-16	98-16	98-16		98-16	06-16	04-16				04-06							
EE01	VILSANDI	95-16	94-16	94-16	94-15	94-16	94-16	94-15	94-16	95-96	-	-	94-16	94-16	-	-	94-16	94-97			94-04		94-16		94
EE02	SAAREJÄRVE	94-16	98-16	94-16	94-16	94-16	94-16	94-15	95-16	95-14	94-16	96	94-16	94-16			96-16	96-12	12		94-15	94-16	96-16	98-14	
ES02	BERTIZ	08-16	08-16	07-16		07-16	08-16	10-15	07-16		07-16		08-16	08-16			07-12	07		07					
FI01	VALKEA-KOTINEN	88-16	94-15	88-15	88-96	89-15	89-99	88-89	89-01		88-16	87-16	88-01	90-97		90-93	88-91	88-09			88-97		90	87-89	87
FI03	HIETAJÄRVI	88-16	93-00	88-15	89-96	89-15	89-99	88	89-01		88-16	87-16	88-01	90-97		90	88-91	90-09			90-97		90-91	87-89	
FI06	PALLASJÄRVI			14-15																				88-89	
IE01	BRACKLOON WOOD			91-16		91-11	92-97		91-16			-	91-96	91-98	-	-									
IT01	RENON-RITTEN	90-16	93-16	93-14		93-13	93-13	93-11	93-13		00-13	-	93-10	00	-	-	92-13	09		05-09	92		93-11		
IT02	MONTICOLO-MONTIGGL	77-13	93	93-14		93-13	93-13	93-10	93-13		-	-	93-01	00	-	-	92-13				92		93-11		
IT03	PASSO LAVAZE	92-08	93-13	92-13		94-13	94-00	93-95	95-07		01-13	-	93-05	94	-	-	93-09	95-09		99-09	92				
IT05	SELVA PIANA	97-08	97-15	97-15		97-15	97-15	95	02-08		-	-	97-05		-	-	97-09	09		99-09					
IT06	PIANO LIMINA	99-08	97-16	97-16		97-16	97-16	95			-	-	97-05		-	-	97-09	09		99-09					
IT07	CARREGA	97-08	97-16	97-16		97-16	97-00	95			98-13	-	97-05		-	-	97-09	09		99-09					
IT09	MONTE RUFENO	97-08	97-16	97-16		97-16	97-00	95	02-08		97-14	-	97-05		-	-	97-09	09		99-09					
IT10	VAL MASINO	97-08	00-15	97-15		97-15		95	05-07		-	-	97-05		-	-	97-09	09		99-09					
IT11	ROTI		97-11	97-12		97-12		95			-	-	97-05		-	-	97-09	09		99-09					
IT12	COLOGNOLE	97-01	97-15	97-15		97-15	97-00	95			-	-	97-05		-	-	97-09	09		99-09					
IT13	LA THUILE	97-08	97-15	09-15		09-15		95			-	-	97-05		-	-	97-09			99-08					
LT01	AUKSTAITIJA	93-13	93-16	93-16	93-10	93-16		93-05	94-12	93-12	93-14		06-16	99-16	12		00-13	93-15		02-15	93-16	93-16			93
LT03	ZEMAITIJA	90-13	95-16	95-16	06-10	95-16		94-05	95-12	95-12	95-14		06-16	99-16	95-12		00-13	94-15		02-15	94-16	94-16			94
NO01	BIRKENES	87-16	87-16	87-16	92	89-17		86	89-17	87-88	87-16	-	86			-	91-03	86-13			86				
NO02	KÅRVATN	87-91	87-16	87-16	88	89-11		89	89-09		87-16	-	89			-	92-03	89-09							
NO03	LANGTJERN		87-97	77-16		86-03			91-03		87-16														
PL01	PUSZCZA BORECKA	16	16	16		16			16			16						16							
PL05	WIGRY	16	16	16		16			16		16			16				16							
PL06	PARSENTA	94-16	16	94-16		96-16			16		94-16			16											
PL07	POJEZIERZE CHELMINSKIE	16	16	16							16														
PL08	KAMPINOS	16	16	16		16		16			16			16				16							
PL09	LYSOGORY	16	16	16		16			16		16			16				16							
PL10	BESKIDY	94-16	16	94-16		02-16			16		94-16			16				16							
PL11	WOLIN	16	16	16					16		16			16											
PL12	ROZTOCZE	16	16	16		16			16		16			16				16							
RU03	CAUCASUS BR	89-94	89-16	89-98																					
RU04	OKA-TERRACE BR	89-06	89-16	89-98	90										93-99		93-14	93-02			93		94-96		
RU12	ASTRAKHAN BR	93-94	93-16	93-94																					
RU13	CENTRAL FOREST BR	93	93-94	93													09-16								
RU14	VORONEZH BR	94	94-16	94-98																					
RU16	VELIKIY ISLAND				89-90			89	89	89						93-99	93-16	91-94			89-94	93	94-95		91
SE04	GÅRDSJÖN F1	87-16	88-16	87-16	95	87-16		95	87-16	79-16	87-16	-	99-16	96-16		-	97-01	95-16	91-15	91-15	96-16	92-16	95-16		
SE14	ANEBODA	96-16	96-16	96-16	95	96-16			95-15	96-16	96-16	-	99-16	95-16		-	97-01	82-16	96-16	06-16	97-12	97-16	95-16		
SE15	KINDLA	97-16	96-16	96-16		96-16		97	95-15	97-16	96-16	-	97-16	95-16		-	98-01	96-16	98-13	98-13	98-13	97-16	95-16		
SE16	GAMMTRATTEN	99-16	99-16	99-16		99-16			00-15	00-16	99-16		99-16	00-16			00-01	99-16	99-14	99-14	00-15	00-16	00-16		
UA01	KARADAG	12-13	12-13																						

Figure 1.1. Geographical location of ICP IM sites.

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2 Progress report on dynamic soil-vegetation modelling

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2.1 Summary of progress

The monitoring and research activities at the sites of the ICP IM, ICP Forests programmes and LTER-Europe network produce high quality data that is valuable for identifying ecosystem and biodiversity responses to combined effects of elevated N deposition and climate change. During previous years monitoring data from selected sites of these networks have been utilized to setup a dynamic modelling study for the evaluation of future vegetation responses to continuing nitrogen deposition. This report summarizes the approach and the progress. The VSD+ model chain (Fig. 2.1) has been applied to simulate soil properties: soil solution pH, soil base saturation (BS) and soil organic carbon to nitrogen ration (C:N). VSD+ calibrations have been finalized (Table 2.1) at nearly 30 sites in ten countries (Austria, Belgium, UK, Germany, Italy, Norway, Poland, Serbia, Sweden and Finland). The PROPS plant model has been tested with long-term data of forest understory vegetation (Fig. 2.2). In a final step the entire model chain will be used to assess current legislative efforts to reduce N deposition by additionally accounting for expected climate change. These results will be used for work within the CLRTAP Working Group on Effects, and in the EU/H2020 project eLTER.

2.2 Results on soil and plant model validation

The dynamic soil model VSD+ (Bonten et al. 2016) has been calibrated at 26 sites. Scenario analysis of deposition and climate change impacts on BS, C:N and pH have been reported in Holmberg et al. 2018. The calibration results are summarized in Table 2.1.

Table 2.1. Measures of performance of dynamic soil modelling

	BS	C:N	pH	$[H^+]$ ($\mu eq\ L^{-1}$)	$[NO_3^-]$ ($\mu eq\ L^{-1}$)	$[NH_4^+]$ ($\mu eq\ L^{-1}$)	$[SO_4^{2-}]$ ($\mu eq\ L^{-1}$)	[Ca+Mg+K] ($\mu eq\ L^{-1}$)
N sites	24	23	26	26	13	8	11	8
N observations	25	34	224	224	171	97	144	100
NMAE[a]	0.23	0.10	0.06	0.41	0.78	0.98	0.60	0.48
Pearson[b]	0.92	0.92	0.94	0.91	0.69	0.29	0.66	0.63
RSqr[c]	0.84	0.84	0.89	0.84	0.47	0.08	0.44	0.39
CE[d]	0.81	0.83	0.86	0.81	0.27	-0.09	0.18	0.37

[a]NMAE Normalized mean absolute error; [b]Pearson correlation coefficient; [c]RSqr Coefficient of determination; [d]CE Coefficient of efficiency

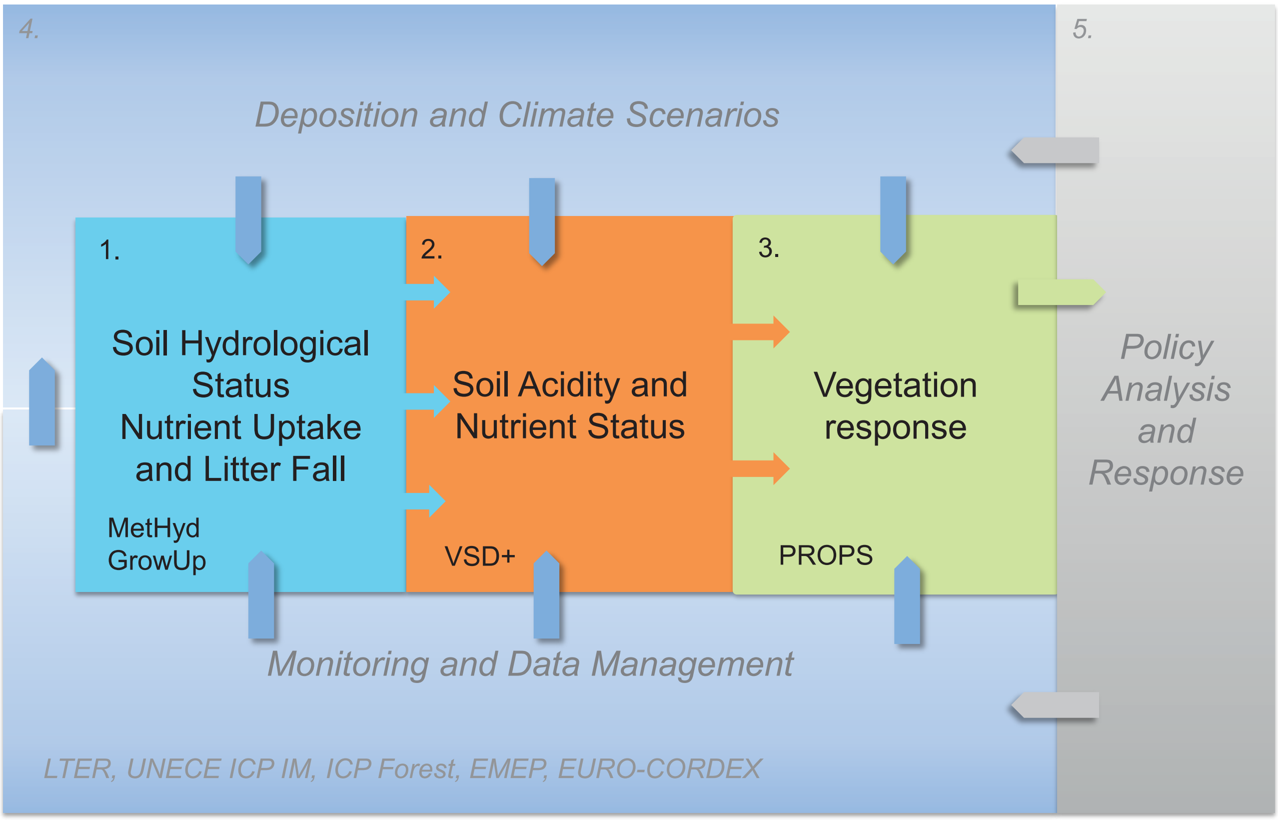

Figure 2.1. The model chain from 1) MetHyd and GrowUp to the dynamic soil model 2) VSD+ simulating soil acidity and nutrient status as an input to the empirical plant model 3) PROPS. Box 4 denotes the supporting components: monitoring and data management infrastructures by the LTER, ICP IM and ICP Forests networks, and EMEP and EURO-CORDEX-related services for providing data on current and projected deposition, and regional climate projections. Box 5 illustrates the use of this system approach in policy support work.

In its current version, the PROPS model is a database holding statistical niche functions for 4053 plant species occurring in Europe that were derived from a huge set of vegetation records together with associated soil data (Reinds et al. 2014). The outputs of PROPS are probabilities of species occurrences as a function of precipitation, temperature, N deposition, soil C:N ratio and soil pH. Long-term records of vascular plant species covering the period between the years 1982 and 2017 from several of these sites were used to compare observed versus modelled changes in forest understory vegetation. This dataset is an extension of the one used in another ICP IM work (Dirnböck et al. 2014). Scientific nomenclature for plants was standardized using the R package "Taxonstand" version 2.1 (Cayuela et al. 2012). The focus of the validation was on indicator species groups useful as biodiversity metrics (Rowe et al. 2016). To calculate biodiversity metrics for oligo- and acidophilic plant species, species-specific indicator values for nitrogen (N) and soil reaction (R) (Ellenberg et al. 1992) were assigned to long-term vascular plant and bryophyte species records. Species with indifferent indicator values were excluded from subsequent analyses. Regional Ellenberg indices were used for Atlantic study plots (Fitter & Peat 1994) and Mediterranean study plots (Pignatti et al. 2005) by using the R package "TR8" version 0.9.18 (Bocci 2015). Species with low Ellenberg (Ellenberg et al. 1992) N and R value (≤4) were deemed oligophilic and acidophilic, respectively.

Here results regarding trends in oligophilic species, i.e. species sensitive to N deposition, are presented. For each study site with an observation interval ≥ 10 years (n=19), temporal change in this group of species were characterized by calculating the mean response ratios as the natural logarithm of the ratio between the first (t_{n-1}) and last observation (t_n), i.e. $RR = ln(Xt_n/Xt_{n-1})$. Similarly, mean RRs were calculated across all study plots. Metaregression analyses were used to test each RR for a significant deviation from zero (metafor R package, Viechtbauer 2017). Of the 28 study plots, four plots were excluded because of only one available vegetation record (DE02, DE03, DE04 and RS02). Some further plots could not be included because of having only one species in this group. Response ratios between observed and modelled metrics were examined with linear regression and RMSE. This analysis resulted in a weak relationship between modelled and measured trends in oligophilic species (Fig. 2.2). Taking into account the many factors affecting plant species changes (overstory tree changes, management, etc.), and the complexity of the model chain, this relationship is considered a reasonable proof for its applicability.

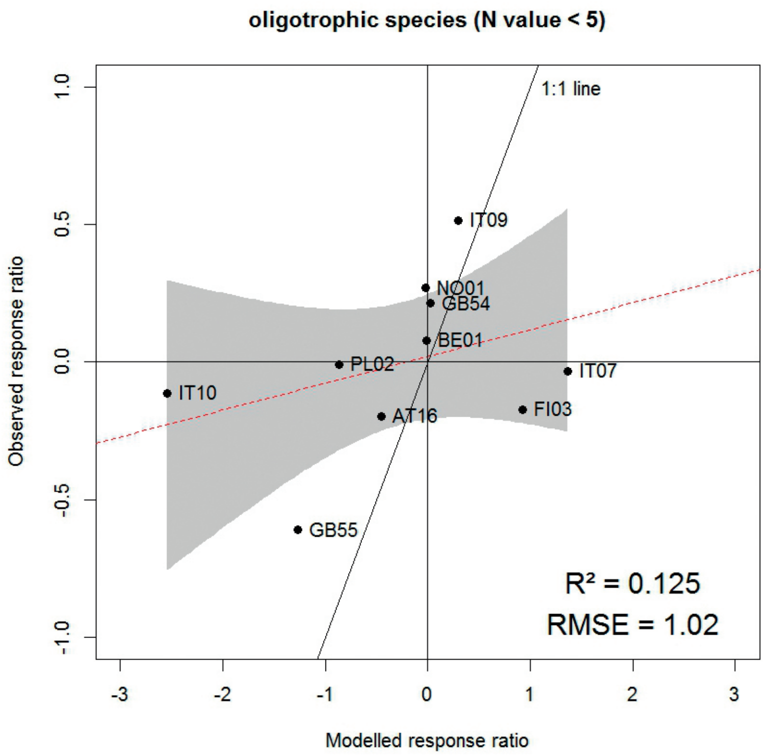

Figure 2.2. Relationship between modelled and measured trend in oligophilic (Ellenberg N value ≤ 4) plant species per site. Trends are expressed as response ratios.

2.3 Outlook

Scenario combinations with two deposition and two climate scenarios (ensemble of 12 climate models each) until 2100 have been processed and the results have been summarized in a scientific manuscript (Dirnböck et al. submitted).

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3 Long-term changes in the inorganic nitrogen output fluxes in European ICP Integrated Monitoring catchments – an assessment of the role of N-related parameters in catchments

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3.1 Introduction

A long-term (1990–2015) pattern of sulphur (S) and nitrogen (N) emission reduction responses in large areas across Europe is shown by trend analysis from the ICP Integrated Monitoring (ICP IM) network of forested research catchments (Vuorenmaa et al. 2018). The emission control programmes have been particularly successful for S, and the concentrations and deposition fluxes of anthropogenic sulfate (xSO_4) decreased significantly ($p < 0.05$) almost at all studied IM sites (Fig. 3.1). Substantially decreased xSO_4 deposition has evidently resulted in a decrease of xSO_4 concentrations and fluxes in runoff in forested catchments in large parts of Europe, as shown at IM sites (Fig. 3.2). The IM catchments have increasingly responded to the decreases in deposition of xSO_4 during the last 25 years, and the most acid-sensitive IM catchments are experiencing a recovery from sulphate-driven acidification, indicated by clear increases in pH and acid neutralizing capacity (ANC) in the soil-water ecosystem. Total inorganic nitrogen ($TIN=NO_3+NH_4$) deposition has decreased in most of the IM areas as well, but to a lesser extent than that of xSO_4. Deposition of NO_3 and NH_4 decreased significantly at 60–80% (concentrations) and 40–60% (fluxes) of the sites, which have resulted in a decrease of TIN in runoff. Concentrations and fluxes of NO_3 in runoff decreased at 73% and 63% of the sites, respectively, and NO_3 concentrations and fluxes decreased significantly at 50% and 21% of the sites, respectively. The ICP IM network covers important deposition gradients in Europe, and these results confirm that emission abatement actions are having their intended effects on precipitation and runoff water chemistry in the course of successful emission reductions in different regions in Europe, even though decreasing trends for S and N emissions and deposition and deposition reduction responses in runoff water chemistry tended to be more gradual since the early 2000s. This study strongly emphasises the importance of the larger scale integrated long-term monitoring and research of different ecosystem compartments for detecting the variety of impacts of changing environmental conditions on ecosystems.

Globally increasing trends in surface air temperature are widely documented, and a significant increase in annual air temperature records was detected at 61% of the IM sites in 1990–2015. The significant increasing monthly trends were detected mostly during spring (April-May) and late autumn (November). Annual precipitation and runoff records showed almost equally positive and negative trend slopes, but trends were rarely significant. The site-specific variation of xSO_4 concentrations in runoff was most strongly explained by deposition. Climatic variables and deposition explained the variation of TIN concentrations in runoff at single sites poorly, and as yet there are no clear signs of a consistent deposition-driven or climate-driven increase in TIN exports in the catchments.

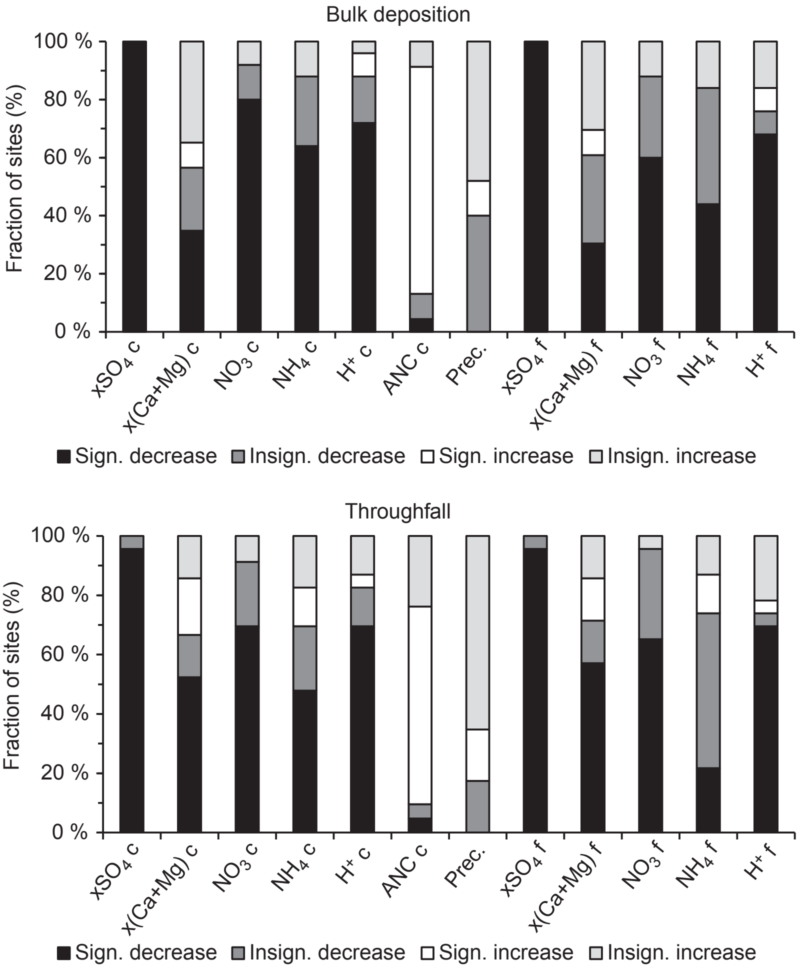


Figure 3.1. Percentage of Integrated Monitoring sites with a significant decreasing (black), insignificant decreasing (dark grey), significant increasing (white) and insignificant increasing (light grey) trend in concentrations (denoted as c) and fluxes (denoted as f) of bulk deposition (top) and throughfall (bottom) in 1990–2015.

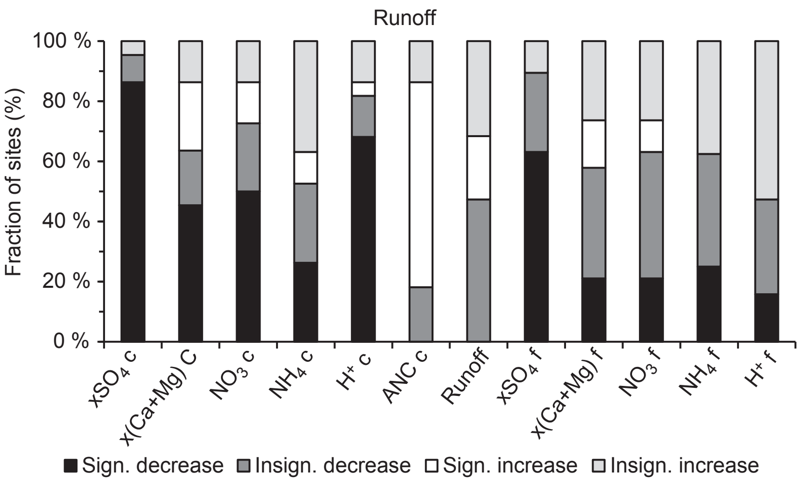
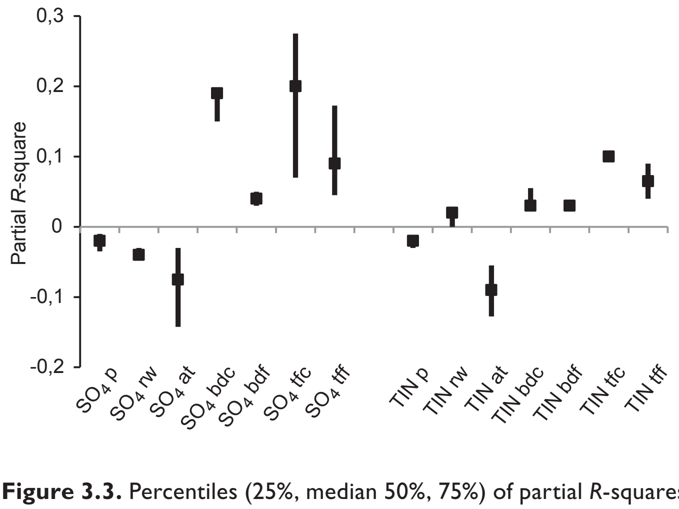


Figure 3.2. Percentage of Integrated Monitoring sites with a significant decreasing (black), insignificant decreasing (dark grey), significant increasing (white) and insignificant increasing (light grey) trend in concentrations (denoted as c) and fluxes (denoted as f) of runoff in 1990–2015.

3.2 The role of N-related parameters in catchments

The present trend of TIN deposition at IM sites is decreasing, which should generally lead to decreased NO_3 concentrations in runoff (e.g. Forsius et al. 2005, Holmberg et al. 2013). The trends for the concentrations and output fluxes of TIN at IM sites are, however, still variable, indicating that surface water-watershed nitrogen dynamics are inherently complex, as nitrogen is strongly affected by biological processes and hydrological conditions, and nitrate concentrations in surface waters may fluctuate greatly by season and spatially across ecosystems. Elevated leaching losses of TIN are generally linked to high N deposition, but losses and trends of NO_3 may be highly variable between sites exposed to relatively similar levels of N deposition. It is obvious that also other factors than TIN deposition – which are not yet fully understood – may largely modify TIN losses and trends from forested catchments.

Long-term trends in the annual input-output budgets of TIN (Vuorenmaa et al. 2017) and monthly runoff water chemistry and fluxes of TIN (Vuorenmaa et al. 2018) in IM catchments were evaluated in relation to changes in emissions and hydrometeorological conditions. Variations of retention/net release of TIN in catchments and NO_3 concentrations in runoff for each of the study sites were explained using multiple statistical analysis. The influence of long-term variation of climatic variables and deposition on TIN concentrations in runoff at single sites did not strongly arise from this data set and analysis (Fig. 3.3), but it is obvious that not all potential drivers were included in the empirical models in these studies. The IM sites are located in areas with very different N deposition gradients, and further analysis with specific catchment data, such as landscape and soil data, is needed to elucidate the variation in inorganic N concentrations at IM sites (see literature review of Weldon in this report regarding the importance of different processes involved). The next phase of the work will be an assessment of the role of N-related parameters in the IM catchments, involving collection and analysis of available landscape data and physical and chemical soil data. The main aims of the study are to evaluate the present status of these internal nitrogen parameters, and to analyse if these parameters explain the variation/trends at IM sites. The national focal points and the representatives for the sites will be invited to assist with these activities.



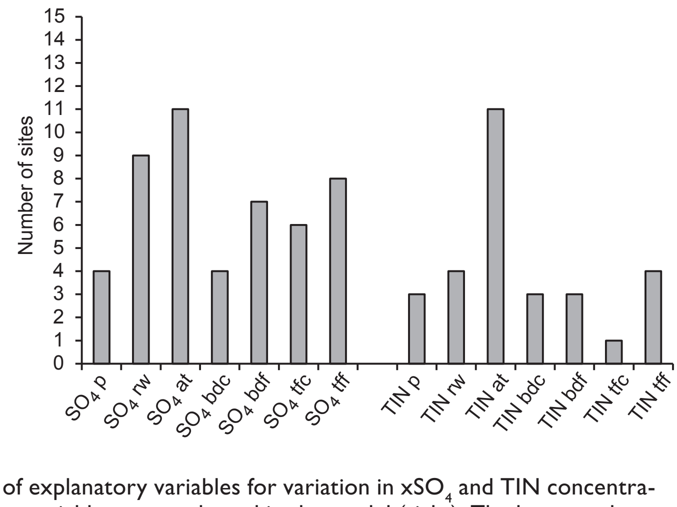


Figure 3.3. Percentiles (25%, median 50%, 75%) of partial *R*-squares of explanatory variables for variation in xSO_4 and TIN concentrations in runoff (left), and number of sites in which different explanatory variables were selected in the model (right). The lower and upper lines indicate 25th and 75th percentiles, respectively, and a square indicates the median value (p, precipitation; rw, runoff volume; at, air temperature; xSO_4 bdc, xSO_4 concentration in bulk deposition; xSO_4 bdf, xSO_4 flux in bulk deposition; xSO_4 tfc, xSO_4 concentration in throughfall; xSO_4 tff, xSO_4 flux in throughfall; TIN bdc, TIN concentration in bulk deposition; TIN bdf, TIN flux in bulk deposition; TIN tfc, TIN concentration in throughfall; TIN tff, TIN flux in throughfall.

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Weldon, J. 2018. Post disturbance vegetation succession and resilience in forest ecosystems – a literature review (this report, chapter 4)

4 Post disturbance vegetation succession and resilience in forest ecosystems – a literature review

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4.1 Introduction

Temperate and boreal forests have in the past often been conceived of as the stable end point of processes of succession (Clements 1916), albeit subject to occasional stand-replacing disturbances that start the predictable and directional process of succession anew (Siren 1955). From the 1970's onwards however, forest-based research focusing on disturbance as an important process creating heterogeneity and structuring communities began, along with a growing awareness of the importance of scale (Turner 2010). It is now recognised that forests are dynamic systems at all stages of succession, that disturbance at different scales plays a key role in their development, and also that they have the potential to develop along multiple successional pathways (Angelstam & Kuuluvainen 2004, Taylor & Chen 2011). Studies at least as far back as Zackrisson (1977) have recognised that fire is an important agent in creating a dynamic mosaic of vegetation in different stages of post-disturbance succession, but one whose impact has been much reduced by active fire suppression in the north European context. Other natural disturbances have maintained or increased the level of their impacts in recent decades however, with wind damage and outbreaks of bark beetle (*Ips typographus*) currently being among the most important disturbance mechanisms influencing the dynamics of unmanaged spruce dominated forests in Europe (Schelhaas et al. 2003, Thom & Seidl 2016). Temperate and boreal forests have of course evolved subject to disturbances such as insect herbivore outbreaks, wind and fire occurring at different scales and intensities, and have developed a certain resilience to them (Gutschick & BassiriRad 2003). Indeed, these disturbances have a fundamental role in shaping the development, structure and function of forest ecosystems, opening gaps and initiating succession processes that improve many indicators of biodiversity (Thom et al. 2016). However, even unmanaged semi-natural forests are additionally subject to diffuse anthropogenic disturbances such as eutrophication and acidification via atmospheric deposition of nitrogen (N) and sulphur (S), and climate change, which have the potential to result in a loss of biodiversity and changes in vegetation community composition (Dirnböck et al. 2014), negatively impacting forest resilience to future disturbance. While much progress has been made in reducing sulphur emissions linked to damaging "acid rain", levels of N deposition remain stubbornly high and are of growing ecological concern (Michel & Seidling 2016, Sutton 2011). It is an open question whether forests can continue to manifest resilience to natural disturbances when additionally stressed by such diffuse anthropogenic disturbance factors, and if so for how long. Decreased resilience may not be immediately obvious, instead taking the form of a resilience debt that manifests only after future disturbances (Johnstone et al. 2016). A natural disturbance such as an insect outbreak which would otherwise come within the resilience capacity of a forest may then suffice to push the ecosystem past a threshold and into an alternative state, with potentially undesirable outcomes for both biodiversity and the provision of ecosystem services (Pope et al. 2014). Research focussed on ecological resilience has generally focussed on single disturbance agents (Donohue et al. 2016) but interactions between disturbances are both common and potentially of critical importance in predicting outcomes (e.g. a warmer climate increases drought stress, reducing the resistance of trees to the attacks of beetles (Bentz et al. 2010) which are in turn able to increase their reproductive rates under warmer conditions (Seidl et al. 2014)). Given that the frequency and severity of natural disturbance agents is predicted to increase under conditions of global change (Seidl et al. 2011) there is a pressing need for a greater understanding of how the combined effects of anthropogenic background disturbances and natural disturbance cycles may impact thus-far resilient ecosystems.

4.2 Resilience theory

Ecological resilience can be defined as "a measure of the amount of change needed to change an ecosystem from one set of processes and structures to a different set of processes and structures" (Angeler & Allen 2016), and was first developed in an ecological context by Holling (1973). It is important to note that this is a fundamentally different concept to the engineering definition of resilience as the time taken for a system to return to its original state after disturbance, as ecological resilience allows for the possibility that a shift to an alternative stable state (sometimes visualised as a basin of attraction) can occur. In such cases the end of a disturbance or perturbation is not enough to return the system to its pre-disturbance state and active management intervention may be required (Holling 1973). Stable states are not static however, and internal dynamics continue (Scheffer et al. 2001).

The concepts of resilience and alternate stable states were further developed with the idea of cycles of adaptive change (Holling 1985), in which systems proceed through the phases of growth (a burst of rapid exploitation of resources), conservation (the slower accumulation of "capital", growth of connections and rigidity, loss of resilience), release (a rapid collapse that releases accumulated energy) and reorganisation (repeating the same cycle or potentially shifting to a new one). These adaptive cycles can be considered as operating in a hierarchy, where the same structure is found at different temporal and spatial scales – a concept labelled a "panarchy" (Holling 2001). The reorganisation stage is a potential point for a shift to a new regime, but whether or not this occurs is influenced by structures occurring at larger scales which can act as a form of conservative ecological memory (Allen et al. 2014).

When considering whether a regime shift is likely, four key attributes of resilience can be identified (Walker et al. 2004): latitude (the width of the basin of attraction), resistance (the depth of the basin of attraction), precarity (how close the system currently is to a threshold), and cross-scale relations/panarchy (how the attributes above in the system of interest are influenced by processes at scales above and/or below). For a regime shift to occur this resilience must be overcome. There are several key factors that make this more likely – the presence of positive feedbacks, the action of multiple causalities, cross-scale interactions, and a high sensitivity to external drivers (Raffa et al. 2008).

If the disturbance is sufficient to escape the current basin of attraction, then an alternate state can be reached, a result (most often) characterised by the dominance of organisms with different life forms, and maintained by stabilising feedbacks based on both biological and non-biological mechanisms (Scheffer et al. 2001).The same reinforcing processes that underlay the resilience to change of the system in its first equilibrium state then contribute to maintaining the system in its new, alternative equilibrium. Returning conditions to the last transition point is no longer enough to affect a regime shift. Instead this requires changing conditions far enough that the other tipping point in the system is reached (Scheffer et al. 2001). Ecological resilience per se then is not an unqualified good, as such undesirable states can also demonstrate resilience to remedial action (Angeler & Allen 2016). Modelling suggests that in a homogenous landscape disturbance either causes a regime shift or there is a full recovery (van de Leemput et al. 2018). In heterogenous landscapes, however, a local disturbance can persist indefinitely in a state of 'no recovery', allowing multiple stable states to co-exist.

Resilience may be compromised by human impacts, such as the removal of functional groups/response diversity, emissions and pollutants that can stress ecosystems (e.g. climate change, N deposition etc.), and changes to the disturbance regime (e.g. through fire suppression in forests) (Folke et al. 2004). In considering the interaction of various disturbances on the resilience of an ecosystem, it is useful to consider temporal scale and distinguish between slow 'press' impacts which operate over longer timescales and fast 'pulse' impacts which have a more sudden effect (Holling 1986, Walker et al. 2004). In the present context it is useful to differentiate the impacts of (in this case anthropogenic) diffuse, slowly developing drivers of change such as N deposition and climate change and the effects of more short lived natural disturbances such as storm damage and outbreaks of herbivorous insects (Thom et al. 2013). A system affected by diffuse disturbances may recover more slowly or not at all from a pulse disturbance (van de Leemput et al. 2018). How such disturbances combine and interact with ecosystem resilience is an important but understudied area (Jessie et al. 2003).

4.3 Nitrogen deposition- a diffuse anthropogenic disturbance

Anthropogenic inputs have vastly increased the total amount of reactive N (i.e. N in forms easily available to living organisms) available globally relative to pre-industrial levels (Galloway et al. 1995). The total anthropogenic input of reactive N was estimated to be 220 Tg N yr^{-1} in 2010, roughly the same amount as all biological N fixation in unmanaged ecosystems (Fowler et al. 2015).

Although considerable progress has been made in reducing levels of sulphur (S) deposition since concern grew about the impact of so-called "acid rain" on forests in the 1980's, levels of N deposition have proven much harder to reduce, and remain an ecological concern (Erisman & De Vries 2000, Michel & Seidling 2016). N limitation on plant growth can be considered the default state of a wide range of temperate and especially boreal ecosystems (Tamm 1991, Vitousek & Howarth 1991) but many forests are no longer N limited as a result of anthropogenic inputs – in the European context while plant growth in low deposition Scandinavian boreal forests is generally still N limited (Hyvönen et al. 2008), high levels of N deposition in central Europe have resulted in some previously N limited forests there becoming limited instead by the availability of phosphorus (Jonard et al. 2015). Within the network of International Cooperative Programme on Integrated Monitoring of Air Pollution Effects on Ecosystems (ICP IM) sites, it is estimated (Holmberg et al. 2013) that even though the number of sites exceeding critical loads for N deposition has decreased since 2000 (with some exceptions in northern and east European sites (Waldner et al. 2014)), at least half of all sites will remain vulnerable to eutrophication under all plausible deposition scenarios over the coming decades. An assessment of total inorganic N across the European ICP IM monitoring site network over 28 years to 2012 found a mixed response in terms of output fluxes despite the general decline in deposition rates, with few significant trends – it appears that deposited N is well retained by sites unaffected by natural disturbances, at least so far (Vuorenmaa et al. 2017).

4.4 Natural sources of N and soil processes

Biological fixation of atmospheric dinitrogen (N_2) is an important source in temperate and boreal forests but can be exceeded by anthropogenic inputs. In boreal Scandinavian forest for example, N fixation (here primarily from associations of feathermoss and cyanobacteria (DeLuca et al. 2002)) is generally smaller than deposition levels, even in the north where deposition levels are considerably lower than the southern areas. Estimates of N fixation rates range from 0.01 to 3.5 kg N ha^{-1} yr^{-1} for boreal forests (DeLuca et al. 2002, Lindo et al. 2013, Stuiver et al. 2015). However, the feathermoss-cyanobacteria contribution to available N drops steeply with increased N deposition as both moss biomass and N fixation rate decline, while their tissue concentrations of N increase. The net effect is that the size of the feathermoss N pool remains stable at levels of simulated N deposition up to 12 kg ha^{-1} yr^{-1}, meaning that they have the potential to buffer the effect of at least a moderate level of atmospheric N (Gundale et al. 2013, 2011).

The other main source of N in boreal and temperate forests is litter decomposition. While this is principally a form of internal cycling of N within the system rather than an external input it is an important process for determining N availability (Parton et al. 2007). Slow decomposition rates due to recalcitrant forms of litter with high levels of lignin or high C:N ratios (e.g. conifer needles, bryophytes) and/or low temperatures can result in the build-up of a large N pool in soil that is largely unavailable to plants (Tamm 1991). N availability for plants is also strongly affected by soil microbial activity. Microbial responses to N depend on the amount of deposition – data from long term monitoring of spruce forests in Germany showed that gross N mineralisation and nitrification increased up to intermediate levels of N deposition and then slightly decreased at higher levels of N enrichment, as microbial biomass/activity is reduced (Corre et al. 2007). Under low N conditions ectomycorrhizal fungi use more N for themselves passing on less to their plant hosts (which makes the common plant response of allocating more C to roots as a response to low N curiously ineffective (Hasselquist et al. 2016, Högberg et al. 2016), but as N supply increases soil bacteria which release more N than fungi do become dominant in the microbial community, opening up the N cycle and potentially facilitating the establishment of nitrophilous plant species (Högberg et al. 2017).

4.5 Negative impacts of N deposition on forests

N deposition is a form of stress that can reduce the resistance of plants to drought, frost, herbivores and pathogens. (De Vries et al. 2014, 2000, Nordin et al. 1998). The soil acidification associated with N deposition can affect mineralisation and result in deficiencies of nutrients, as base cations are lost, and Al is released (which has a toxic effect particularly on root growth and further inhibits take-up of base cations, as well as being toxic to stream fauna) (De Vries et al. 2014, 2000)

There are also direct physiological impacts on plant metabolism (Stulen et al. 1998), provoking shifts in C allocation towards the canopy within trees while roots are much less affected (potentially increasing vulnerability to storm damage (De Visser 1994)). A similar shift from below ground to above ground C allocation with increasing N has also been seen in understory plants (Hasselquist et al. 2016). The eutrophication related increase in tree canopy biomass relative to root biomass also increases vulnerability to drought, and increases the demand for nutrients (Ca,Mg,K,P) the uptake of which is hindered by higher levels of dissolved NH_4 in soils (Bobbink et al. 2002, Erisman & De Vries 2000). In addition, levels of essential nutrients in soils may be inadequate for the increased requirements of the faster growing trees fostered by N deposition and higher levels of atmospheric CO_2, resulting in nutrient deficiencies (Jonard et al. 2015). N accumulation can also result in species change (discussed more fully below).

The severity of these impacts depends on a range of factors including duration, the amount and form of deposits, the sensitivity of the affected habitat, abiotic conditions (e.g. soil acid neutralising capacity, nutrient availability, nitrification potential, N immobilisation rate), and the management history of the site (Bobbink et al. 2002).

Gilliam's (2006) review identifies the processes of vegetation change in forests that are mediated by changes in the availability of N- herbivory, competition, mycorrhizal infection, disease and invasive species – and suggests a hypothesis of increasing homogenisation in forest vegetation. Under N limited conditions, N availability is spatially heterogeneous, and so plant diversity is increased, whereas under chronic deposition of high levels of N cause both to become more heterogeneous. Also, initial N deposition would increase the importance of light limitation in forests (by reducing the effect of N limitation) but further N deposition impacts directly on the competitiveness of plant species.

4.6 Evidence of vegetation changes

Numerous studies have demonstrated N mediated changes in forest floor vegetation at smaller scales, although before reviewing these findings it is important to note some limitations of the methods most commonly employed – experimental additions and observations across spatial/temporal gradients. It has been argued that the high dose rate N additions and short timescales typical of N addition experiments are a poor guide to long term vegetation responses to chronic deposition, as the rate of addition is at least as important as the cumulative total (Binkley & Högberg 2016). On the other hand, in observational studies across spatial and/or temporal gradients it can be very challenging to disentangle the effects of N deposition from other causal factors (Binkley & Högberg 2016).

Strongly N limited sites (and acidic soils (Simkin et al. 2016)) are particularly vulnerable to N deposition induced vegetation changes, while N rich sites are effectively buffered by the small size of N depositions compared to the large existing N pool (Diwold et al. 2010). High levels of background deposition can also make it difficult to provoke changes with experimental N additions – Gilliam, Hockenberry & Adams (2006) explain the lack of response in a temperate hardwood forest by referring to the 10 kg N ha^{-1} yr^{-1} wet N deposition, a suggestion supported by Hedwall et al. (2013), who found a reduction in the effect of N addition in experiments performed in higher N deposition areas.

The threshold for species composition change in European forests is now considered to be 10–15 kg ha^{-1} yr^{-1} (Achermann & Bobbink 2003, Bobbink & Hettelingh 2010, Bobbink 2004, Bobbink et al. 2002) but in low deposition boreal areas additions as low as 6kg N ha^{-1} yr^{-1} can have a significant impact (Nordin et al. 2005) on vascular plant community composition (it is also worth noting that epiphytic lichens are extremely sensitive to N pollution, with a critical load as low as 2.4 kg ha^{-1} yr^{-1} (Giordani et al. 2014).) The suggested critical load for boreal forests is therefore lower, at 5–10 kg ha^{-1} yr^{-1} (Bobbink & Hettelingh 2010). Eutrophication effects on forest floor vegetation community have been found at sites which exceed critical loads (Dirnböck et al. 2014). Increased N generally increases the amount of forest understory cover, particularly facilitating graminoids at the expense of dwarf shrubs (Bobbink et al. 2010).

Bryophyte composition can change particularly strongly with additional N, with some species being highly sensitive (Bobbink et al. 2002, Nordin et al. 2005), and these changes can be very long lasting, even with no further N inputs (Bobbink et al. 2002). In Scandinavian coniferous forests, increased N typically causes an increase in *Deschampsia flexuosa* and decreases in the dwarf shrubs *Vaccinium myrtillus* and *Vaccinium vitis idaea* (Strengbom et al. 2003, Strengbom et al. 2002). This change is partly caused by increased vulnerability to pathogens in *V. myrtillus*, inducing defoliation which allows more light to reach competing *D. flexuosa* (Nordin et al. 1998, Strengbom et al. 2002), a species which is better able to make use of additional N (Nordin et al. 2006, 2005).

Evidence that N deposition can lead to an increased homogeneity of understory vegetation was found using circa 20 years of data from a monitoring site in Austria receiving around twice (28–43 kg ha^{-1} yr^{-1}) the critical load for eutrophication effects (Hülber et al. 2008). An analysis of the Swedish national forest inventory data from 1994–2013 also demonstrated a eutrophication effect, seen in the decline in dwarf shrubs and increase in mesotrophic forbes and ferns found in the temperate zone (where deposition levels are around 12.5 kg N ha^{-1} yr^{-1}) but not in the boreal zone (deposition levels around 2 kg ha^{-1} yr^{-1}) after allowing for the potentially confounding effects of management changes (Hedwall & Brunet 2016).

Despite the above results showing eutrophication effects at stand and landscape scales, it has proven difficult to detect broad-scale (e.g. European level) N deposition effects on forest floor vegetation comparable to the negative effects of N deposition on grassland species richness which have been demonstrated at a very large scale (Stevens et al. 2010). One potential confounding factor in unmanaged forests is that N deposition may result in increased canopy growth (Jonard et al. 2015) limiting nutrients available to understory plants but also affecting light availability. Attempts to find an N driven response in understory plants should take into account changes in the overstorey (Binkley & Högberg 2016), as under closed canopies light is often the most important limiting factor (Gilliam & Roberts 2003).

A meta-analysis of N addition studies worldwide found a clear negative relationship between the cumulative dose of N and species richness over a range of vegetation types (and varied soil and climatic conditions) when considered together but failed to show such a relationship in forest understory taken separately (de Schrijver et al. 2011). Using Europe-wide ICP Forests data, an Ellenberg value based study found some limited evidence of eutrophication effects (Seidling & Fischer 2008) but further analysis of data from the same source did not find any consistent directional change in vegetation over time (Seidling et al. 2008). A different statistical approach based on a partly overlapping dataset also showed a surprisingly small effect attributable to anthropogenic N deposition (van Dobben & de Vries 2010), given the strong effects often seen in smaller scale investigations.

Other studies have also shown mixed results. A study of semi-natural European temperate woodland understory using long time scales found significant shifts in species composition but no directional change that could be attributed to N deposition (Verheyen et al. 2012), probably due to the confounding effects of changes in canopy density affecting light availability. A meta-analysis of temperate forest resurvey studies (Bernhardt-Römermann et al. 2015) found no impact on diversity that could be linked to N deposition during the study period, but some evidence that earlier N deposition had shifted baselines by the time of the first surveys. Van Dobben and de Vries (2016) however used relatively short-term data from managed forests to show that changes in vegetation were significantly correlated to N deposition, alongside an increase in mean Ellenberg N value, although other factors such as soil type explained more of the observed variation.

While Dirnböck et al. (2014) found no Europe-wide trend across all sites, by focusing on sites where the critical load is exceeded, significant relationships emerged between increased N deposition above CL levels and decreasing cover of oligotrophic plants, and a higher proportion of nitrophilous plants were found among newly occurring species in surveys. These conclusions have been questioned (Binkley & Högberg 2016) for taking insufficient account of the influence of possible changes in the overstorey, despite the fact that the authors restricted their analyses to plots that experienced relatively weak change of the tree layer in order to minimise the possible confounding effect of changes in light regime. A large scale response to N deposition in forest understory was also found in a recent analysis of vegetation data (herbaceous plants only) from across the United States which demonstrated a unimodal response to N deposition across vegetation types, with increased species richness at low N levels, then decreasing richness above certain levels (13.4 kg ha^{-1} yr^{-1} for forest) (Simkin et al. 2016).

While small scale experimental studies have clearly shown that N addition can result in changes in forest understory vegetation, finding evidence of similar effects of N deposition at the largest scales has proven more difficult due to confounding factors. Nevertheless, some recent analyses have shown convincing evidence of changes that can be attributed to N deposition, and further investigations of such large-scale impacts are needed.

4.7 Natural pulse disturbances

Bark beetles are major agents of disturbance in temperate and boreal forests. Half of all biotic damage to European forests in the 1950–2000 period can be attributed to bark beetles (primarily *Ips typographus*) and the damage caused by outbreaks increased by +2.5% per year during the second half of the 20th century (Schelhaas et al. 2003).

A healthy conifer has formidable defences against bark beetle attack but recently windthrown trees allow beetles to colonise at low densities, building up their overall numbers while avoiding intraspecific competition until a large scale outbreak can occur (Eriksson et al. 2005). Disturbances such as storms clearly facilitate bark beetle outbreaks in this way, with most being preceded by disturbance in the four years preceding the outbreak (Pasztor et al. 2014).

Once underway, an outbreak proceeds quickly – at a storm damaged site in southern Sweden, initial wind damage took the proportion of dead spruce from 22% to 32% but in the subsequent years the bark beetle attack increased this by around 5% per year after virtually all spruce with a diameter greater than 5 cm were colonised and killed (Löfgren et al. 2014).

It is worth noting that wind damage has increased by +4.2% per year during the second half of the 20th century (Schelhaas, Nabuurs & Schuck 2003), an important increase in disturbance in itself as well as increasing the number of felled trees that are potential breeding areas for bark beetles. Wind damage is not the only disturbance that can potentially facilitate beetle outbreaks however – drought can also cause a sudden increase in the number of stressed trees that are potential hosts, allowing an outbreak to occur (Raffa et al. 2008). Root rot fungi (*Heterobasidion* sp.) can make trees more vulnerable to both beetle attack and windthrow (Hertert et al. 1975, Whitney et al. 2002). It is possible that other forms of disturbance that result in stress to trees also contribute to outbreak occurrences, including N deposition (Eatough Jones et al. 2004).

These disturbances can also have a strong influence on various biogeochemical processes such as carbon, water and nutrient cycling (Raffa et al. 2008). The impact on ecosystem N cycling is of particular interest here and varies strongly depending on deposition levels and vegetation response. A storm followed by a beetle outbreak at a Swedish site saw a long term but modest increase in NH_4 and NO_3 concentrations in water samples due to the low level of N deposition in the area, N uptake by vegetation growing rapidly in gaps created and storage in woody debris (Löfgren, Grandin & Stendera 2014). A similar spruce dominated site in Germany affected by an outbreak of comparable severity saw much higher levels of N leaching however, reflecting the higher N deposition in the area (Zimmerman et al. 2000).

These pulse disturbances are usually seen entirely negatively from an economic perspective, but they can also play a positive role by creating (among other effects) large amounts of dead wood. Many endangered species are reliant on dead wood (Stevens et al. 2010), and the diversity of insects benefits strongly from the effects of a bark beetle outbreak (Müller et al. 2008). They also create stand-level gaps, change the distribution of different classes of tree size and age, and alter the understory composition, accelerating succession. At a landscape scale they create or alter the mosaic of different stand ages, structures and plant communities, which in turn shape the progress of future disturbances (Raffa et al. 2008).

4.8 Expected succession after disturbance

Although forest management by humans has become the most important agent of disturbance in most European forests and has a decisive role in determining forest structure and function (Gauthier et al. 2015) that is not to say that natural disturbances are unimportant, having significant impacts on both managed forests and in semi-natural forests. Different disturbance regimes can be identified working at different scales – from gap dynamics (frequent, low severity, small scale disturbances) to stand replacing disturbance that initiates succession processes, with intermediate disturbances affecting patch and cohort dynamics (Angelstam & Kuuluvainen 2004). A review of disturbance regime studies in Fennoscandian boreal forest showed the long but highly variable time scales needed for a forest affected by a stand-replacing disturbance to move to a regime dominated by patch/gap dynamics – from 100 to over 300 years depending on environmental variables (Kuuluvainen & Ankala 2011).

Early successional ecosystems after a stand replacing disturbance can show increased diversity, as survivors, opportunists and specialists exploiting new niches co-exist (Swanson et al. 2011). This increase in diversity and/or complexity can persist a surprisingly long time. Surveys have found that at least the first 25 years of succession after a bark beetle outbreak demonstrate a significant increase in vascular plant species diversity and a shift towards nitrophilous and light demanding species, with a prolonged early-seral phase providing structural complexity (Winter et al. 2015). This complexity may be a form of 'precocious development' (Donato et al. 2012) with characteristics associated with old growth forests visible on a smaller scale during post-disturbance succession (e.g. vertically heterogeneous canopies with gaps, the presence of both shade tolerant and intolerant species and a large amount of dead wood). A small percentage of mature spruce usually survive a major disturbance, providing a seed source (Kupferschmid & Bugmann 2005) and facilitating the eventual regeneration of a forest with a very similar species composition to that found pre-disturbance (Nováková & Edwards-Jonášová 2015) while many understory plant species can persist through disturbances either as established plants, seeds or rootstocks (Swanson et al. 2011). Such processes can be conceived of as a form of conservative ecological memory enhancing system resilience (Jõgiste et al. 2017, Johnstone et al. 2016).

Modelling of forest change in central Europe under predicted climate and disturbance regimes suggests that an increased frequency and severity of disturbance can reduce the time taken to regain a stable state (Thom et al. 2017). However increased disturbance size had the opposite effect, possibly due to the impact of increased dispersal distances. Disturbance also has scale dependant impacts on biodiversity – large disturbances increase biodiversity on a landscape scale but can have negative impacts on a smaller scale, where increased homogenisation can result, and small isolated populations of species of concern may be threatened. This raises the question of how large a protected area needs to be in order to see these benefits rather than risk only negative impacts from a large scale disturbance event, with the suggestion that 10,000 Ha (of coniferous montane forest) should suffice to ensure high habitat heterogeneity persists (Lehnert et al. 2012). It is not just biodiversity that is of concern during succession processes however, and the provision of ecosystem services by forests must also be considered. A review of the impact of wind, fire, and bark beetle disturbances in boreal and temperate forests on ecoservices provision and biodiversity reveals a so-called "disturbance paradox" – disturbances generally have a positive effect on biodiversity but a negative effect on the provision of ecosystem services (Thom & Seidl 2016). In terms of carbon storage as an ecosystem service for example, the average reduction in C storage after disturbance is 38.5%, while the average increase in species richness is 35.6%. This is a problem since 90% of the world's temperate/boreal forest is neither strictly protected nor solely used for forestry, instead being expected to be multi-functional.

4.9 Disturbance leading to a new stable state?

Combined disturbances may be simply cumulative but may also interact in surprising ways, with the first disturbance altering the characteristics of the second to create a novel situation that provokes non-linear responses, overcoming forest resilience. A study in US subalpine coniferous forest adapted to fire found, for example, that earlier storm damage created the necessary preconditions for a later fire to have unusually long duration, creating very large disturbed patches which are potentially moving to new stable states dominated by deciduous trees (Buma & Wessman 2011). Interacting disturbances do not necessarily reduce resilience however – thinning or thinning and burning treatments simulating moderate natural disturbances have been shown to increase forest resilience to further perturbations such as bark beetle attacks (Hood et al. 2016), while other interacting disturbances are a part of normal forest dynamics (storm damage facilitating beetle outbreaks for example). Disturbances which are likely to provoke a shift to a new state are likely to be i) novel, ii) of increased frequency, intensity or extent, or iii) a combination of interacting disturbances (Trumbore et al. 2015), although the origin of disturbances in categories i) and ii) is also likely to involve disturbance interactions, such as climate change related drought increasing fire frequency (Hanson & Swanson 2001). Repeated examples of the same disturbance type may either improve resilience or overcome it depending on the interval (Davies et al. 2009), while two disturbance types can produce different patterns of regeneration in the same conifer forest and a third regeneration pattern when combined (Johnstone et al. 2016). Seidl et al. (2017) concluded that disturbances will be amplified by interactions between different disturbance factors and that coniferous forests are likely to be most affected. The complex interactions between changing disturbance regimes and forest resilience have the potential to result in serious and widespread impacts on forest ecosystems, whether the principle concern is biodiversity or ecosystem services, yet a satisfactory understanding of these mechanisms is yet to be achieved. The careful monitoring of succession processes at sites which have undergone severe combined disturbances including anthropogenic stresses may indicate regime shifts and be a useful step in furthering our understanding of these mechanisms.

In 2004, Angelstam and Kuuluvainen's review of European boreal forest disturbance and successional dynamics concluded that "Little is known, however, about the interaction between different disturbance factors in the natural forest" (Angelstam & Kuuluvainen 2004), in 2010 disturbance interactions were still "poorly understood" (Turner 2010), and as recently as 2015 the urgent need for an improved understanding of the relationships between interacting stresses/disturbances and forest health at different scales was stressed (Trumbore et al. 2015). Much remains to be done.

4.10 Conclusions

Disturbance regimes and associated patterns of regeneration and succession are important parts of forest dynamics and can help to foster biodiversity and resilience. However, the additional stresses placed on forest ecosystems by diffuse anthropogenic impacts act to reduce that resilience. At the same time, the natural disturbance regime to which forests are adapted is intensified by anthropogenic factors. This combination has the potential to induce regime shifts in forest ecosystems that negatively impact biodiversity and the provision of ecosystem services. Numerous experimental studies have shown that N inputs are capable of causing major changes in forest vegetation and some recent studies have provided evidence that ongoing N deposition is indeed changing understory vegetation and affecting canopy growth. The combined effects of N deposition then, are likely to result in lower forest resilience in ecosystems simultaneously facing an increased frequency and/or intensity of natural disturbances including storm damage and outbreaks of bark beetles. Such disturbance interactions can have unpredictable and surprising consequences which are as yet insufficiently studied, and sites that have experienced severe combined perturbations may show evidence of regime shifts.

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Annex 1

Report on National ICP IM activities in Sweden in 2016

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Introduction

The Swedish integrated monitoring programme is run on four sites distributed from south central Sweden (SE14 Aneboda), over the middle part (SE15 Kindla), to a northerly site (SE16 Gammtratten). The long-term monitoring site SE04 Gårdsjön F1 is complementary on the inland of the West Coast and has been influenced by long-term high deposition loads. The sites are well-defined catchments with mainly coniferous forest stands dominated by bilberry spruce forests on glacial till deposited above the highest coastline. Hence, there has been no water sorting of the soil material. Both climate and deposition gradients coincide with the distribution of the sites from south to north (Table 1). The forest stands are mainly over 100 years old and at least three of them have several hundred years of natural continuity. Until the 1950's, the woodlands were lightly grazed in restricted areas. In early 2005, a heavy storm struck the IM site Aneboda, SE14. Compared with other forests in the region, however, this site managed rather well and roughly 20–30% of the trees in the area were storm-felled. In 1996, the total number of large woody debris in the form of logs was 317 in the surveyed plots, which decreased to 257 in 2001. In 2006, after the storm, the number of logs increased to 433, corresponding to 2711 logs in the whole catchment. In later years, 2007–2010, bark beetle (*Ips typographus)* infestation has almost totally erased the old spruce trees. In 2011 more than 80% of the trees with a breast height over 35 cm were dead (Löfgren et al. 2014) and currently almost all spruce trees with diameter of ≥20 cm are gone.

Table 1. Geographic location and long-term climate and hydrology at the Swedish IM sites.

	SE04	SE14	SE15	SE16
Latitude; Longitude	N 58° 03′; E 12° 01′	N 57° 05′; E 14° 32′	N 59° 45′; E 14° 54′	N 63° 51′; E 18° 06′
Altitude, m	114–140	210–240	312–415	410–545
Area, ha	3.7	18.9	20.4	45
Mean annual temperature, °C	+6.7	+5.8	+4.2	+1.2
Mean annual precipitation, mm	1000	750	900	750
Mean annual evapotransporation, mm	480	470	450	370
Mean annual runoff, mm	520	280	450	380

In the following, climate, hydrology, water chemistry and some ongoing work at the four Swedish IM sites in 2016 are presented (Löfgren 2017).

Climate and Hydrology in 2016

In 2016, the annual mean temperatures were higher (0.4–1.4 °C) compared to the long-term mean (1961–1990) for all four sites. Largest deviation occurred at the northern SE16 site. Compared with the measured time series, 16 years at site SE16 and 20 years at the other sites, the temperatures in 2016 were somewhat higher at all the IM sites. These values were slightly lower than in 2014 and 2015 when temperatures were the highest observed for the whole measurement period with exception for SE15 Kindla where the temperature was slightly higher in the years 1999 and 2000. The variations between years have been considerable, especially for the last five years, over 3 °C at three of the sites. Smaller variations were found at the central site SE15 Kindla, only 1 °C.

Compared to the long-term average values (1961–1990), the precipitation amounts in 2016 were much lower at three of the sites with up to 171 mm at SE14 Aneboda corresponding to 77% of the long-term average. For SE16 Gammtratten in the north, this value was 131 mm (81 %) and at the central site SE15 Kindla, it was 71 mm (92%). Only at site SE04 Gårdsjön, the precipitation amount was on average. Especially the summer and autumn precipitation was low.

The characteristic annual hydrological patterns of the catchments are for the southern sites high groundwater levels during winter and lower levels in summer and early autumn. In 2016 at SE14 Aneboda, high groundwater levels were observed in February and thereafter successively lowered until late autumn when the aquifers started to be refilled. For SE16 Gammtratten in the north, snowmelt occurred in April–May with the highest groundwater levels in the end of May. After that the water level was subsiding until next spring. At site SE15 Kindla, a more varying pattern was observed with several peaks 0.2 m below the soil surface during February to May. Summer rains also created groundwater peaks, but the lowest levels, close to one meter below soil surface, were found in late summer. The aquifer was refilled in November. The patterns were fairly similar to those in 2015. The groundwater level has decisive influence on the discharge values (Fig. 1).

In addition to precipitation, evapotranspiration affects the runoff pattern. In 2016, the runoff pattern was fairly typical for SE16 Gammtratten with a marked snowmelt peak in May. The sites SE04 Gårdsjön and SE15 Kindla showed fairly typical patterns during the first 6 months followed by unusually low discharges in July to October due to low precipitation. In November, ordinary discharge levels occurred. For SE14 Aneboda low discharge was observed from April/May throughout the rest of the year, reflecting the very low discharge and lake water levels found in the southern part of Sweden this year (Fig. 1).

At the two northern sites, generally snow accumulates during winter and the groundwater levels stay low furnishing low discharge. However, warm winter periods with temperatures above 0 °C have during a number of years contributed to snowmelt and excess runoff also during this season. As a consequence, the spring discharges have been comparably low during snowmelt, deviating from the normal conditions. In 2016, the central and northern sites SE15 Kindla and SE16 Gammtratten, respectively, showed relatively normal winter and spring runoff patterns (Fig. 1).

In 2016, the annual runoff made up 31–83% of the annual precipitation, which was a wide range compared to the ordinary 40–60% found in previous years. The highest share was found at the northern site SE16 Gammtratten (83%), due to a rather intense snowmelt period and fairly cold climate during the rest of the year, yielding low evapotranspiration (17%) and high runoff (Table 2). At site Aneboda (SE14),

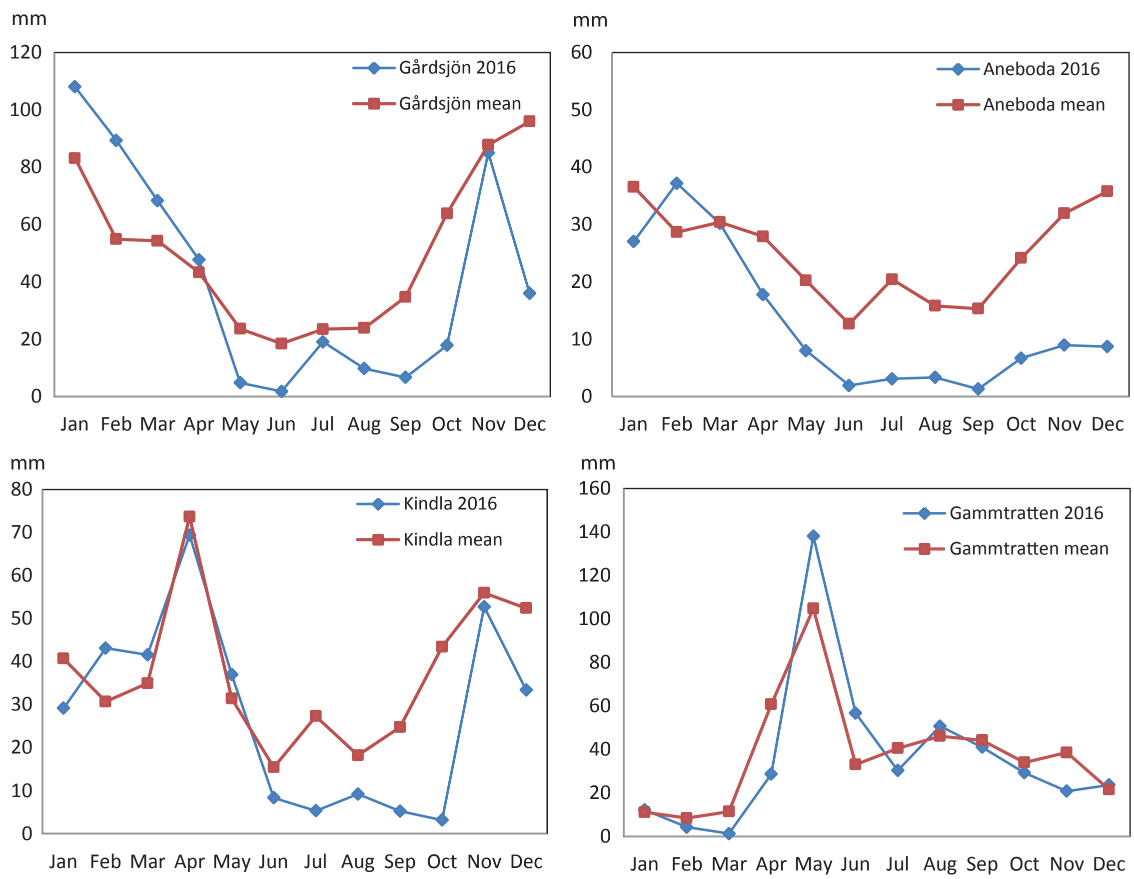

Figure 1. Discharge patterns at the Swedish IM sites in 2016 compared to monthly averages for the period 1996–2016 (mean). Note the different Y-axis scales.

storm felling, followed by bark beetle attacks, have reduced the forest canopy cover, inducing low interception. Actually, the measured throughfall reached 89% of precipitation. The total evapotranspiration was estimated to 349 mm, a value lower than in the previous years.

At SE04 Gårdsjön, evapotranspiration and runoff were equally large, each constituting 50% of the precipitation (Table 2), which deviates from earlier years when runoff often made up 2/3. At SE15 Kindla, the annual runoff was 337 mm, which was fairly low compared to the 30 years long-term mean of 450 mm and compared to the mean of the monitoring period of 449 mm (Table 2). Low precipitation with 823 mm resulted in 486 mm evapotranspiration, i.e. 69% of precipitation, a rather high value despite a high throughfall on 655 mm.

Table 2. Compilation of the 2016 water balances for the four Swedish IM sites.
P – Precipitation, TF – Throughfall, I – Interception, R – Water runoff

	Gårdsjön SE04		Aneboda SE14		Kindla SE15		Gammtratten SE16	
	mm	% of P	mm	% of P	mm	% of P	mm	% of P
Bulk precipitation, P	975	100	503	100	823	100	525	100
Throughfall, TF	757	78	449	89	655	80	557	106
Interception, P–TF	218	22	54	11	168	20	-32	-4
Runoff, R	494	51	154	31	337	41	436	83
P–R	481	49	349	69	486	59	89	17

Water chemistry in 2016

Low ion concentrations in bulk deposition (electrolytical conductivity = 1–2 mS m^{-1}) characterise all four Swedish IM sites. The concentrations of ions in throughfall, including dry deposition, were higher at three sites. At SE16 Gammtratten, the conductivity in throughfall (1 mS m^{-1}) was almost the same as in bulk deposition indicating very low sea salt deposition and uptake of ions by the trees. At the two most southern sites, sea salt deposition provides tangibly higher ion concentrations, especially at the west coast SE04 Gårdsjön site (6.5 mS m^{-1} in throughfall). The catchments groundwater pathways are fairly short and shallow, providing rapid surface water formation from infiltration to surface water runoff. However, conductivity in the soil water was higher compared to throughfall, showing influences from evapotranspiration and soil chemical processes. The acidity in deposition has during the last 10 years been rather similar at all sites with somewhat higher pH values (0–0.5 units) in throughfall compared with bulk deposition. However, in 2016 SE04 Gårdsjön had a throughfall pH on 5.1 while the two sites SE14 Aneboda and SE15 Kindla had values on 5.3 (Table 3). For SE16 Gammtratten, pH values were 4.8 and 4.9 in bulk deposition and throughfall, respectively.

Table 3. Mean deposition chemistry values 2016 at the four Swedish IM sites. S and N in kg ha^{-1} yr^{-1}.

	SE04	SE14	SE15	SE16
pH, bulk deposition	4.9	4.9	5.4	4.8
pH, throughfall	5.1	5.3	5.3	4.9
S, bulk deposition	3.5	1.8	1.3	0.8
N, bulk deposition	7.6	6.7	3.9	2.2

During the water passage through the catchment soils, organic acids were added and leached to the stream runoff. In the upslope recharge areas, pH in the upper soil layers (E-horizon) was mainly lower than in throughfall. However, in the peat in discharge areas at SE15 Kindla and SE16 Gammtratten, pH was higher compared to throughfall while it was similar to throughfall at SE14 Aneboda, but considerably lower at SE04 Gårdsjön with a pH of 4.4. In the recharge areas, the buffering capacity in soil water and groundwater varied between negative and positive values, but were most frequently on the negative side, especially for SE15 Kindla. In the discharge areas, the buffering capacity in groundwater was fairly high with ANC exceeding 0.14 mEq L^{-1} at SE14 Aneboda and SE15 Kindla and with bicarbonate (HCO_3^-) present at Aneboda, Kindla and Gammtratten at average concentrations of 0.02, 0.14 and 0.20 mEq L^{-1}, respectively. At SE04 Gårdsjön ANC was lower (0.03 mEq L^{-1}). The stream waters were acidic with pH values below 4.8 at all sites except Gammtratten having a pH of 5.6. The stream water buffer capacity was positive at all sites, even though SE04 Gårdsjön and SE15 Kindla had an ANC close to 0 mEq L^{-1}. Anions of weak organic acids contributed to the positive ANC and bicarbonate contributed at SE16 Gammtratten.

The share of major anions in deposition was similar for sulphate, chloride and nitrate at three of the sites, while chloride dominated at SE04 Gårdsjön due to the proximity of the sea. In throughfall, organic anions contributed significantly at all four sites. The chemical composition changed during the catchment soils passage and the sulphate concentrations were higher in stream water compared with deposition, indicating desorption or mineralization of previously accumulated sulphur in the soils. At Aneboda, nitrification contributed to fairly high nitrate values in the recharge area soil water (0.03–0.46 mEq L^{-1}), but was considerably lower in the discharge areas, probably due to nitrogen uptake and denitrification.

Besides effects on ANC and pH, the stream water chemistry is to a considerable extent influenced by organic matter. At Aneboda (SE14), the DOC concentration was high with 24 mg L^{-1} while the other sites Gårdsjön (SE04), Kindla (SE15) and Gammtratten (SE16) showed lower values 13, 11, and 10 mg L^{-1}, respectively. High DOC concentrations create prerequisites for metal complexion and transport as well as high organic nitrogen fluxes. The organic nitrogen concentrations in stream water ranged from 0.20 to 0.63 mg N L^{-1}. The shares of Norg/Ntot were 70–90%, showing Norg dominating Ntot, and with SE14 Aneboda having the lowest share and SE16 Gammtratten and SE15 Kindla on the highest range. Inorganic nitrogen (NO_3-N and NH_4-N) was low at three sites with <0.48 mg L^{-1} but higher at SE14 Aneboda with 119 mg L^{-1}, possibly due to the forest damage.

Total phosphorus (Ptot) in bulk deposition varied between 2 µg L^{-1} and 16 µg L^{-1} with the highest values at SE14 Aneboda. In stream water, SE14 Aneboda also showed the highest Ptot (33 µg L^{-1}) as well as DOC concentrations. The other sites had average Ptot concentrations between 3 µg L^{-1} and 9 µg L^{-1} with the highest value at SE16 Gammtratten.

Inorganic aluminum (Al_i), toxic to fish and other gill-breathing organisms, has been analyzed in soil solution, groundwater and surface waters at the IM sites. Relatively high total Al concentrations occurred in the soil solution (0.4–1.8 mg L^{-1}) as well as in stream water (0.25–0.50 mg L^{-1}) at the southern sites Aneboda and Kindla with low pH (ca 4.8). At the northern site SE16 with a pH of 5.6, the total Al concentrations were low, approximately 0.25 mg L^{-1}. Inorganic Al made up 16–46% of the total Al at the three sites (data from 2016 lacking for Gårdsjön), corresponding to 0.04–0.26 mg Al_i L^{-1} with high Al_i at low pH, and the 0.04 mg Al_i L^{-1} at the northern site Gammtratten with higher pH. According to the SEPA classification system, the Al_i concentrations at Aneboda and Kindla are considered extremely high and high at Gammtratten. The priority heavy metals Pb, Cd and Hg were still accumulating in the catchment soils, while the stream concentrations were low compared with the levels causing biological effects. However, methyl mercury, only measured at Aneboda, was still relatively high creating prerequisites for bioaccumulation. In stream water Hg-tot concentration was 8.2 ng L^{-1} with Hg-methyl on 2.2 ng L^{-1}.

In summary, the four Swedish IM sites show low ion contents and permanently acidic conditions. In stream water, only the northern site SE16 Gammtratten had buffering capacity related to bicarbonate alkalinity. Organic matter has an impact on the water quality with respect to colour, metal complexation, and phosphorus concentrations at all sites, but less at SE15 Kindla, where rapid soil water flow paths provide low DOC and acidic waters. For SE14 Aneboda, the forest dieback provides a relatively high share of water runoff as well as high nitrate concentrations compared with the other three sites. SE04 Gårdsjön is strongly influenced by the sea.

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The Integrated Monitoring Programme (ICP IM) is part of the effect-oriented activities under the 1979 Convention on Long-range Transboundary Air Pollution, which covers the region of the United Nations Economic Commission for Europe (UNECE). The main aim of ICP IM is to provide a framework to observe and understand the complex changes occurring in natural/semi natural ecosystems.

This report summarizes the work carried out by the ICP IM Programme Centre and several collaborating institutes. The emphasis of the report is in the work done during the programme year 2017/2018 including:

- A short summary of previous data assessments
- A status report of the ICP IM activities, content of the IM data base, and geographical coverage of the monitoring network
- A report on long-term changes in the inorganic nitrogen output fluxes in European ICP Integrated Monitoring catchments – an assessment of the role of internal nitrogen parameters
- A progress report on dynamic soil-vegetation modelling
- A literature review – Post disturbance vegetation succession and resilience in forest ecosystems
- National Reports on ICP IM activities are presented as annexes

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